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July 1994

Evaluating the Effectiveness of Forestry Best Management Practices in Meeting Water Quality Goals or Standards

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Introduction

Purpose

The purpose of this document is to help forest managers and their staffs develop water-quality monitoring plans to evaluate the effectiveness of forestry best management practices (BMP's) in meeting water-quality goals or standards in the United States. The term "water quality" refers to the chemical, physical, biological, and habitat condition of streams. Approaches and methods described here can also be used to evaluate the effectiveness of grazing, mining, and other nonpoint source (NPS) best management practices.

The document is a reference, not a national protocol. It deals with the design of monitoring projects and the selection of variables and methods for monitoring them given (1) the designated beneficial use, (2) the type and intensity of management, (3) the stream/ecosystem setting, and (4) monitoring objectives or questions related to NPS pollution. The methods apply only to streams of the 4th order and smaller that can be waded during low-flow periods.

With a few exceptions, forest management affects naturally occurring water-quality variables rather than in the introducing unnatural pollutants. This document describes monitoring projects designed to isolate the effect of forest management from other nonpoint sources of pollution and determine whether BMP's or management activities meet NPS goals or standards. BMP's are practices designed to meet water quality goals or standards. Some forest treatments have not been identified as BMP's and are called management activities in this report.

The specific objectives are to: (1) ensure that monitoring projects consider all factors influencing NPS pollution, (2) design monitoring systems that isolate the impacts of BMP's or management activities, (3) review variables to consider when monitoring BMP effectiveness, (4) summarize representative methods appropriate for monitoring NPS variables, and (5) provide some examples of BMP effectiveness monitoring and decisions.

The document is intended for State foresters, managers of national forests, forest industry, State water quality agencies, State conservation agencies, universities, consultants, and any other organizations or individuals involved in evaluating the effectiveness of forest management in meeting water-quality goals or standards.

[redacted]

Background

In 1972 Congress passed the Federal Water Pollution Control Act (P.L. 92-500), commonly called the Clean Water Act. The Clean Water Act's goal is to restore and maintain the chemical, physical, and biological integrity of the Nation's waters. An interim goal of the Act is the protection and propagation of fish, shellfish, and wildlife. Propagation includes the full range of biological conditions necessary to support reproducing populations of all forms of aquatic life and other life that depend on aquatic systems (EPA 1990). EPA's response to this Act is an ecological approach to water-quality monitoring, including monitoring habitat condition.

Under this Act, States have established water-quality standards for streams and other water bodies. Standards vary among States, but consist of a designated beneficial use, criteria to protect the beneficial use, and an antidegradation policy statement.

The States were to adopt narrative biological criteria into their water-quality standards during the period 1991–93 (EPA 1990). EPA expects numeric biological criteria to be adopted in the next few years and is in the process of refining its guidance for biological criteria for streams (EPA 1991). EPA has issued guidance for monitoring the physical, chemical, and biological integrity of streams (MacDonald et al. 1991, Plafkin et al. 1989, and EPA 1990). The EPA guidance clearly demonstrates an aquatic ecosystem approach to standards and monitoring, including the use of integrated physical, chemical, and biological assessment techniques.

BMP's effectiveness monitoring will determine if BMP's protect water quality, that is, yield water that does not contribute to degradation of water quality in the receiving stream. BMP's will be designed, implemented, monitored, and revised (if needed) to assure protection of water quality. In order to evaluate the effectiveness of forestry BMP's, many factors need to be considered.

Recent EPA guidance for monitoring the physical, chemical, and biological integrity of water will directly influence monitoring in forest streams, including the selection of reference sites (streams), which are assessed to develop a reference condition that will provide an attainable measure of ecological health. The reference condition defines the range and variability of physical, chemical, biological, and habitat conditions. It is against the reference condition that the health of other streams is compared.

A stream/ecosystem classification system is needed to ensure proper comparison of study streams with reference streams. Many factors must be evaluated to determine if forest management is impacting water quality and if water quality goals or standards have been met.

Most NPS responses to forestry practices occur during high rainfall or snowmelt runoff events, yet impacts may occur during low-flow periods. Suspended sediment, pesticides, and other pollutants are discharged from the stream system during high streamflow. However, some pollutants settle out during waning stages of runoff and accumulate in the streambed—for example, fine sediments, soil aggregates, and attached pollutants.

Forest management should protect the designated beneficial use: namely cold- or warm-water fisheries; recreation; and municipal water supply; or potentially endangered, threatened, or sensitive aquatic species. Usually, cold- and warm-water fisheries will be the most limiting beneficial use. States are developing biological criteria, including habitat, for protecting this beneficial use. To ensure comparability of present and future monitoring data with standards, forestry should use the same monitoring and analysis methods as the State.

By definition, NPS's are diffuse, coming from the landscape in a dispersed fashion over the surface or through subsurface flow. Therefore, for forest NPS, monitoring methods must determine if what is measured or observed in the stream is influenced by BMP's or management activities on the land. This will require on-land monitoring to develop the linkage. Part of this analysis should explain why in-stream conditions change or remain unchanged.

On-land monitoring includes: (1) evaluating the implementation of BMP's, (2) evaluating the effectiveness of BMP's in reducing erosion and sediment leaving the slopes, (3) measuring the amount of solar radiation through the riparian canopy, (4) determining if pesticide drift has reached the stream, and (5) determining if logging slash has entered the stream. Models, such as the Universal Soil Loss Equation, can be used to compare on-slope erosion between treatments or BMP's and help explain in-stream responses.

Monitoring in-stream conditions above and below a management area might reveal a change in aquatic life and habitat. Does this change fall within the range defined by the reference condition? Is this change really linked to management on the slope or is it natural variation in abundance of aquatic life and habitat quality along the stream? Is it a result of stream-channel morphology changes, effects of prior land use, etc.? Some of these questions can be answered by proper selection and classification of reference streams within ecosystems. The rest must be addressed in a monitoring project designed to link on-slope activities with in-stream changes in physical, chemical, and/or biological variables.

To sum up, effectiveness monitoring answers the questions: Did we get the desired result? Did we protect water quality? Did BMP's meet specific water-quality criteria? Have BMP's been selected, implemented, and maintained properly?

Finally, the accepted practice in water-quality assessment is to use the term "parameter" to refer to chemical and physical constituents of water. In this book, the term "variable" is used in place of "parameter" because many channel morphology, habitat, etc., variables are added to the list of potential measurements to determine the physical, chemical, biological, and habitat conditions. It is best to call all these measurements "variables." However, some excerpts from methods will use the term "parameter."

Planning Monitoring Projects

Effectiveness monitoring in this document seeks to answer five questions:

1. Do properly selected, implemented, and maintained BMP's meet water-quality goals or standards? (Various levels of BMP can be installed and monitored to assess in-stream physical, chemical, and biological responses.)
2. Does a management activity jeopardize water quality?
3. Which management system best protects water quality?
4. How long do BMP's remain effective?
5. How effective are BMP's under major climatic events?

The acceptance or rejection of the null hypothesis is basic to all five questions. Some examples of null hypotheses include:

- BMP's produce water meeting temperature criteria.
- Benthic invertebrates populations below logged areas with BMP's are the same as benthic invertebrate populations in reference streams.
- There is no difference in aquatic habitat between the study stream segment and reference stream.
- All management activities with BMP's protect water quality equally.

The basics for monitoring project planning will be discussed first, followed by some comments appropriate to each of the five questions.

MacDonald et al. (1991) provides a good summary of key steps for designing and implementing water-quality monitoring projects:

> *Although the definition of specific steps in developing a monitoring projects tends to vary according to the author of the guidelines and the particular monitoring situation, the key steps are as follows:*
>
> - *propose—together with the managers—the general objectives;*
> - *define the approximate budget and personnel constraints;*
> - *review existing data;*
> - *determine monitoring parameters, sampling locations, sampling procedures, and analytic techniques;*
> - *evaluate hypothetical or real data;*
> - *reassess monitoring objectives and compatibility with existing resources;*
> - *initiate monitoring activities on a pilot basis;*
> - *analyze and evaluate data;*
> - *reassess monitoring objectives and compatibility with existing resources;*
> - *modify monitoring project as necessary;*
> - *continue monitoring;*
> - *prepare regular reports and recommendations.*
>
> *Figure 1 is a schematic representation of these key steps, and it also indicates some of the critical feedback loops in developing and implementing a water quality monitoring project. In most cases, however, the key steps are not nearly as distinct and sequential as indicated in Figure 1. Decisions*

made at each step often have repercussions for the entire monitoring project, and sometimes this may force a reassessment of previous steps. For example, preliminary identification of the possible sampling locations may necessitate a review of the budget constraints or monitoring objectives. Hence the feedback loops shown represent only the most critical pathways, and each step may not always be completed in the order indicated. What is essential is that each key step be explicitly addressed, and the sequence indicated in Figure 1 is one approach to optimize the process of developing a monitoring project.

The first step is to identify the general objectives, and this best done by the managers in consultation with the technical staff. Once the general objectives have been determined, the approximate personnel and budgetary constraints must be specified in order to ensure that the subsequent monitoring plan is realistic. The availability of past data also must be assessed. If past data are available, it may be possible to evaluate changes over time provided the same measurement techniques and sampling locations are employed. If past data are unavailable, changes probably will have to be assessed by site comparisons, and this often leads to greater flexibility in the selection of both the monitoring parameters and the sampling locations.

The next step is to formulate the specific objectives. This requires the participation of both the managers and the technical staff in order to ensure that the specific objectives are technically and financially feasible. The importance of this interaction is often overlooked, and a failure in communication can lead to a variety of problems. For example, if the manager is unaware of the potential benefits of the monitoring project, obtaining the necessary resources to carry out the project may be difficult. Alternatively, if the technical specialist does not listen to the manager, the specialist may design a monitoring project that will not provide the necessary guidance for management decisions. Input from both the managers and the specialists is needed to balance the need for more data and the cost of acquiring the data. Both sides also must be explicitly aware of the risks and uncertainties associated with monitoring in a highly variable environment.

Often the technical specialist will need to take the lead role in formulating the specific objectives because the specialist will be more familiar with previous monitoring efforts and the likely impacts of management activities on water quality and aquatic resources. Formulation of specific objectives also requires some knowledge of the fluvial system to be monitored and the likely impact on management activities.

Careful identification of the specific objectives probably is the most crucial step in the entire process, as a set of precise objectives will largely define the remainder of the monitoring project, including the approximate cost, monitoring parameters, sampling locations, sampling frequency, and data analysis techniques.

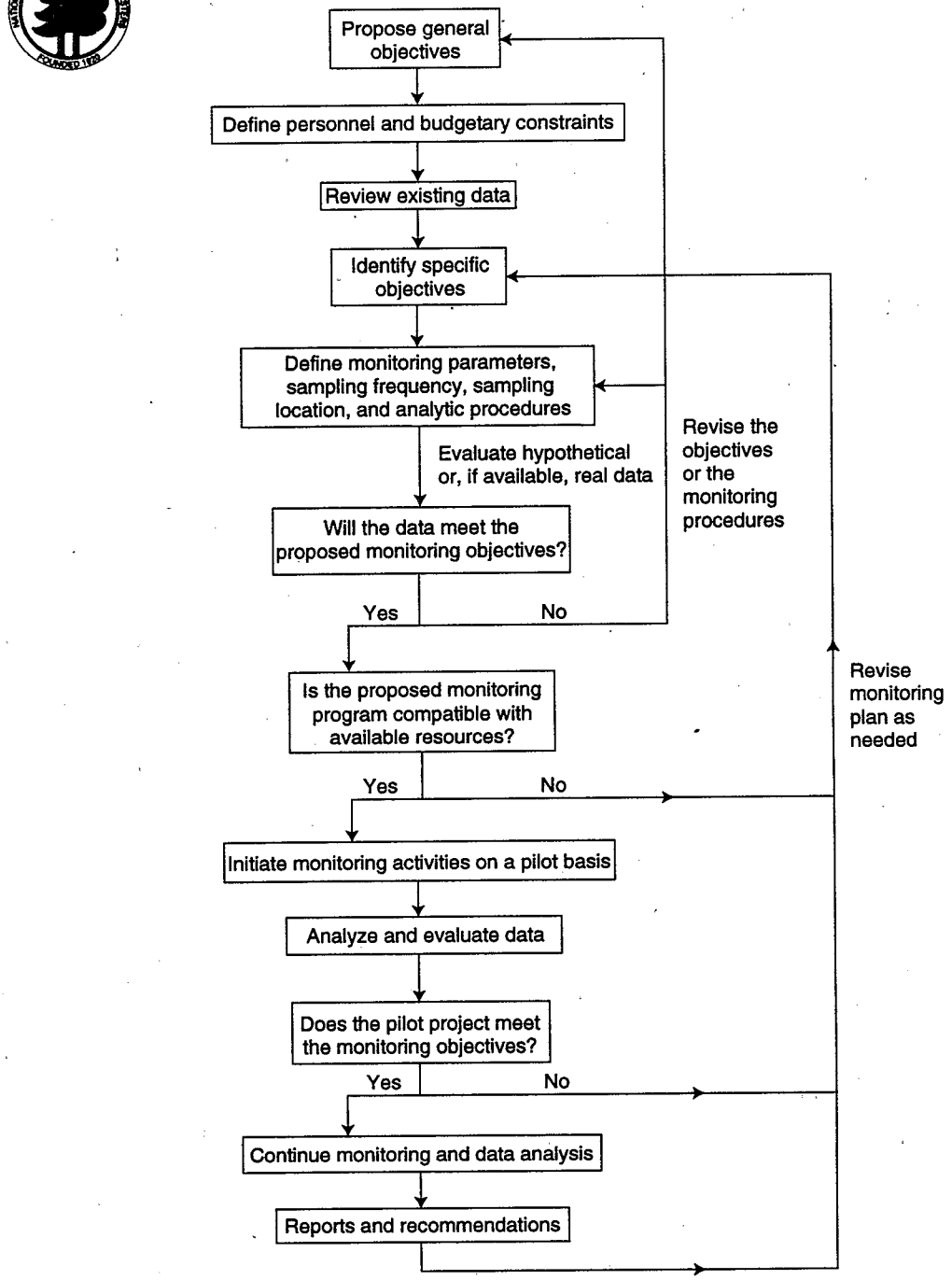


Figure 1. Development of a monitoring project.

Once the specific objectives have been formulated, the next step is to select the parameters to be measured and set out the protocols for collecting data and analyzing field samples. Provision for outside analyses and repetitive samples is needed for quality assurance and quality control. The frequency, duration, and location of measurements will be determined by the objectives and the decisions with regard to the trade-offs among sample size, variability, risk, and uncertainty.

Probably the best means to evaluate the feasibility of the objectives is to develop and test a set of hypothetical or—if available—real data. This is rarely done, but it can be extremely helpful in terms of crystallizing the procedures and attainable objectives. Problems at this stage may necessitate a rethinking of the objectives, a change in the parameters to be monitored, or alterations in the sampling design. If the data are consistent with the other components of the monitoring plan, a final check should be made to ensure that the resources are available to carry out the work, and that responsibilities for each aspect of the monitoring plan are clearly defined.

If the specific objectives are determined to be feasible, the next step is to obtain a final cost estimate in terms of staff time, equipment, and outside expenditures. Delaying the final cost estimates until this step is unusual, but the advantages are (1) the managers already have bought into the monitoring project by helping to define it, and this makes it easier to obtain the necessary support; and (2) the monitoring objectives play a more prominent role in designing the monitoring plan, rather than the monitoring plan being primarily a function of the available staff and expertise. Thus, as indicated in Figure 1, the balancing of monitoring needs and budgetary constraints should be a two-step, iterative process. The first step is simply to ensure that the objectives and scope of the monitoring plan are generally realistic with regard to the available personnel and budget. From that point, however, the planning process should emphasize the optimal achievement of the monitoring objectives. A final synthesis occurs when the monitoring plan has been fully conceptualized.

If at this stage the proposed monitoring plan substantially exceeds the available resources, it may be necessary to revise the monitoring objectives. Alternatively, a smaller reduction in costs might be achieved by reducing the number of sampling sites, reducing the number of parameters to be monitored, or reducing the frequency of sampling. The danger of adjusting sampling intensity rather than the objectives is that the expectations may remain unchanged while the capability or sensitivity of the monitoring project is reduced. By having managers participate in the planning process, they will be much more aware of how additional personnel and budget constraints will alter the anticipated results of the water quality monitoring project.

At this point the proposed monitoring project is ready for data collection to begin. Generally it is best to consider the first field season or set of data collection activities as a pilot project. This allows more flexibility to adapt the methodology to the conditions and variability found in the field. It also provides more impetus to the rapid analysis of field data and subsequent

modifications of the monitoring plan. All too often the monitoring plan is considered as a final, fixed document, and then there is not as much incentive to analyze the data as they are collected. In such cases the data tend to simply accumulate, and it is not until the end of the project that somebody recognizes that the efficiency and quality of data collection could be improved, or that the original monitoring objectives cannot be fully achieved. Designation of the first phase of data collection as a pilot project greatly enhances the potential for communication among all those involved in the monitoring project—technicians, statisticians, managers, and technical specialists.

As shown in Figure 1, the results of the pilot project can lead either to a revision of the monitoring project or to continued monitoring. In most cases a pilot project, if properly formulated, will result in some modifications in the monitoring procedures, but will not alter the basic structure or objectives of the overall monitoring project. Continued monitoring will then lead to the accumulation of data that must be checked, stored, and analyzed. A description of these steps is beyond the scope of this document, but data storage and retrieval is another key aspect of monitoring that is often neglected in the planning phase.

The final step in Figure 1 is the preparation of reports and recommendations. For a variety of reasons many monitoring projects do not follow through to this step, and in such cases the worth of conducting the project must be questioned. In general, the multiple demands on staff time mean that the monitoring data will be used only if they are summarized and interpreted. If the results are clearly presented, the information will be much more widely disseminated, and this will reflect favorably on those responsible for the monitoring project. More importantly, the data are more likely to be evaluated by managers and used for the original purpose, namely the guidance of management decisions. Failure to follow through to this final step implies a basic failure in achieving the objectives of a monitoring project.

Monitoring a watershed is analogous to a person getting a physical exam. During the physical, the doctor takes a blood sample, which is analyzed for sugar, hormones, cell counts, cholesterol, etc. Like the circulatory system is to a human, the stream is to a watershed. Based upon the blood sample, the doctor can judge whether the person is healthy or not. Similarly, a water-quality sample from a stream can indicate the health of a watershed.

If the doctor finds the cell counts out of balance, she runs other tests in an attempt to pinpoint the source of the problem. The doctor uses multiple tests to make a diagnosis before starting treatment.

The same kind of careful efforts are needed to diagnose whether BMP's are effective. The stream sample may reveal adverse impacts on fish, but further investigation needs to pinpoint the source of the problem. All pertinent factors associated with the stream and management practices need to be examined. Table 5 (page 58) identifies various methods to monitor reference and study streams: biology, habitat, channel geomorphology, aquatic vegetation, chemical and physical attributes, and on-slope conditions. All factors need not be

evaluated in every situation; however, all factors should be considered for evaluation. Enough of them must be evaluated to reach a good diagnosis.

The scheduling of monitoring depends on when information on BMP effectiveness can be most effectively and precisely determined. Monitoring can be scheduled within a season, within the period of a management activity, or over a longer period. Often, it is most appropriate to collect water-quality samples before, during, or immediately after a management activity. For example, suspended sediment from constructing a road stream-crossing should be monitored during construction. Herbicide monitoring should be done during the application and/or through the next series of major storms. Seasonal differences can also influence the scheduling of monitoring; for example, monitoring during low-flow periods is best done when dissolved oxygen is being assessed for BMP effectiveness. Benthic macroinvertebrate diversity and abundance are seasonally sensitive, and the best season to compare biological responses will vary by region.

The timing of on-land, channel, or biological assessments can vary with the region and hydrologic/geomorphic processes being evaluated. Monitoring of on-land BMP effectiveness for surface erosion in the South must be conducted soon after treatment because of rapid recovery of sites, usually 2 to 4 years. On the other hand, monitoring the effect of large wood in road fills on landslides would not be appropriate until the woody material has decayed and the road has been exposed to a "testing" storm.

Following are comments on planning monitoring projects for each of the five effectiveness monitoring questions previously stated. Again, it is critical to monitor factors that can link in-stream observations with management activities for all five effectiveness questions.

QUESTION 1. Do properly selected, implemented, and maintained BMP's meet water quality goals or standards?
This question can be approached in two ways. First, specific BMP's can be installed adjacent to study streams. This allows design criteria, such as varying streamside management zone widths or varying percent of canopy retained, to be tested. A BMP design and implementation phase would be required. Similar reference and study streams would have to be preselected.

The second approach locates management areas where BMP's have been or are being installed and maintained properly. These tracts can be located through implementation monitoring. Potential areas to be monitored must be screened to ensure that upstream natural and/or human influences will not mask potential water-quality responses to the activities in question. Upstream conditions should approximate reference conditions. Such screening requires some time, but should be faster than planning and implementing management activities. However, under this alternative, it would be difficult to evaluate BMP design criteria.

QUESTION 2. Does a management activity jeopardize water quality?
When concern about a specific management activity is expressed, a quick monitoring effort is required to determine if water quality is being impaired and, if

needed, assurance that remedial action is prompt. Above and below monitoring sites are commonly used. The stream segment above the management area must be assessed to determine if it approximates reference conditions or if it is heavily impacted by human activities that would mask any impact from the management area in question. Several stream sites above are assessed to determine the range and variability of biological and habitat conditions, which are then compared to the reference conditions, if available. If reference condition data are not available, the least disturbed reference sites are located, assessed, and analyzed to define a reference condition.

Several sites below the management area are monitored to determine if there is impairment compared to the upstream (reference) conditions. The number of sites above and below is determined by statistical needs to make an inference as to whether there are significant differences.

If the State has defined reference conditions and associated water-quality criteria, monitoring stream segments adjacent to and below the management area will provide data for comparison with reference condition and established criteria.

QUESTION 3. Which management system best protects water quality?
In many States, BMP's do not prescribe such management activities as the type of site preparation to use for specific watershed conditions or the type of logging system to use. When these management activities are implemented, some BMP's may be installed—for example, streamside management zones may be left between a prepared site and the stream. The question becomes, which management treatment with associated BMP's best protects water quality?

Planning monitoring projects to answer this question can take the same two approaches as for question 1. First, several management treatments with associated BMP's are planned, implemented, and monitored near streams. These treatments are replicated to define range and variability of responses. The second approach is to locate a range of existing treatments with proper implementation of associated BMP's, find replicates, and monitor to define range and variability of responses. In both approaches, comparisons are made with reference conditions.

QUESTION 4. How long do BMP's remain effective?
Question 4 poses difference challenges. Once installed, do BMP's remain effective through the recovery period of the treatment area? If not, how long do they last and under what climatic conditions? The answers to these questions would help define BMP maintenance programs.

Again, there are two approaches to this question. One approach is to implement a treatment, install associated BMP's, and monitor in-stream and on-slope variables until the treatment area stabilizes. In some regions, this may take just a few years, while other regions may recover very slowly, requiring more than 10 years. The latter requires an extended commitment of time and resources and the answer to questions would be slow in coming.

The second approach is to locate management activities with properly installed BMP's of various ages near streams. Treatment areas should range from new to the age when the treatment area would be stabilized. In-stream and on-slope monitoring are planned to determine trends in recovery and in the effectiveness of BMP's over time. This approach yields answers much sooner than the first approach. The difficulty is determining if BMP's were properly installed on the old treatments. This problem can be partially overcome by having access to good implementation monitoring records or by inspecting treatment areas.

QUESTION 5. How do major climatic events affect BMP effectiveness? Answering this requires a flexible, opportunistic approach to monitoring. It is not practical to install BMP's and hope that a major rainfall or snowmelt event will occur during a short recovery period. Therefore, the monitoring project design must take advantage of major climatic events that occur within reasonable travel distance. The plan would be to go to the affected area, find treatment areas with properly installed BMP's, and monitor associated streams and reference streams. This type of monitoring determines if in-stream responses are associated with management and associated BMP's, with the climatic event alone, or a combination of the two. On-slope monitoring will determine if BMP's survived the storm, or if they were effective in minimizing impacts on streams.

When one plans to monitor the effectiveness of BMP's in meeting water-quality goals or standards, the components monitored must be related to protecting the designated beneficial use or those set as water-quality criteria by the State to protect the beneficial use and can be affected by forest management. Poorly planned monitoring will sample variables that are not related to beneficial uses or water-quality criteria or are not affected by forestry. There is an ongoing effort to develop biological and habitat metrics to use in monitoring water quality. Such metrics will vary by region and the aquatic community, so local knowledge and research need to be used in designing monitoring projects. The State water-quality agency is a good source of this information.

MacDonald et al. (1991) discussed variables most commonly related to a major designated beneficial use (Table 1) and those commonly affected by forest management activities (Table 2). These variables were first defined for the Pacific Northwest and Alaska but are generally applicable to the rest of the country. The sensitivity ratings have been revised to reflect national conditions. These sensitivity ratings are very general and local conditions may cause widely different sensitivities. The parameters and associated sensitivity identified in Tables 1 and 2 are for general guidance and will vary greatly among ecoregions. For example, the sensitivity of pH to roads is rated as "indirectly affected and not very sensitive" in Table 2. However, if roads are cut through rock formations containing iron pyrite, exposing the pyrite to rain will result in formation of acid runoff, which can significantly lower stream pH.

MacDonald et al. (1991) did not address site preparation for tree planting, so site-preparation columns are added to Table 2. These tables can help in selecting variables to monitor, but selection must always be tempered by knowledge of local conditions and management activities.

Table 1. Qualitative assessment of the effects of water quality parameters on the major designated uses of water from forested watersheds in the Pacific Northwest and Alaska. 1 = designated use is directly related and highly sensitive to the parameter in almost all cases; 2 = designated use is closely related and somewhat sensitive to the parameter in most cases; 3 = designated use is indirectly related and not very sensitive to the parameter in most cases; 4 = designated use is largely unrelated to the parameter; V = relationship between the parameter and the designated use is highly variable

	Designated uses affected by water quality parameter						
Water quality parameters	**Domestic water supply**	**Agricultural water supply**	**Hydroelectric generation**	**Recreation**	**Warm-water fishes**	**Cold-water fishes**	**Biological integrity**
Water column							
Temperature	3	4	4	2	2	1	1
pH	1	1	4	3	3	3	3
Conductivity	1	1	4	4	4	4	4
Dissolved oxygen	2	3	4	2	1	1	1
Intergravel DO	4	4	4	3	2	1	1
Nitrogen	2	2	4	2	3	3	2
Phosphorus	2	2	4	2	3	3	2
Herbicides and pesticides	1	1	4	2	3	3	1
Flow							
Peak flows	4	4	1	3	3	2	2
Low flows	2	1	1	2	2	2	3
Water yield	2	1	1	3	4	4	4
Sediment							
Suspended	1	1	1	2	2	2	1
Turbidity	1	2	1	1	1	1	1
Bedload	3	3	2	3	2	2	2
Channel Characteristics							
Cross-sections	4	4	4	3	3	3	3
Width/width-depth ratio	4	4	4	2	2	2	2
Pool parameters	4	4	4	2	1	1	2
Thalweg profile	4	4	4	3	2	2	3
Habitat units	4	4	4	3	1	1	2
Bed material							
Size	3	4	4	3	1	1	1
Embeddedness	4	4	4	3	2	1	1
Surface vs. subsurface	4	4	4	4	2	2	2
Large woody debris	4	4	4	2	1	1	2
Bank stability	3	3	3	2	2	2	2
Riparian							
Riparian canopy opening	4	4	4	2	2	2	1
Riparian vegetation	4	4	4	2	2	2	1
Aquatic organisms							
Bacteria	1	3	4	1	4	4	4
Algae	2	3	4	1	2	2	1
Invertebrates	4	4	4	3	1	1	1
Fish	4	4	V	1	1	1	1

Table 2. Sensitivity of water quality parameters to management activities, assuming average management activities: 1 = directly affected and highly sensitive; 2 = moderately affected and somewhat sensitive; 3 = indirectly affected and not very sensitive; 4 = largely unaffected

	Forest management activities								
			Applications						
Parameter	Harvest	Road building & maintenance	Fertilizer	Herbicides	Pesticides	Intensive site prep.[1/]	Low int. site prep.[2/]	Grazing	Recreation
Water column									
Temperature	1-2	3	4	3	4	1-2	1-2	2	4
pH	3	3	3	3	4	2-3	2-3	3	4
Conductivity	3	3	3	3	4	3	3	3	4
Dissolved oxygen (DO)	2-3	3	2	3	4	3	3	1	4
Intergravel DO	2	2	3	3	4	1-2	2-3	2	3
Nitrogen	2	3	1	3	4	2	2	1	3
Phosphorus	2	3	1-3	3	4	2	2	1	3
Herbicides and pesticides	4	3-4	4	1	1	3	3-4	4	3
Flow									
Peak flows	1-2	1	4	3	4	1	1-2	3	3
Low flows	1	3	3	3	4	1	1	2	4
Water yield	1	3	3	3	4	1	1	3	4
Sediment									
Suspended	1-3	1	3	3	4	1-2	1-3	2	3
Turbidity	1-3	1	3	3	4	1-2	1-3	2	3
Bedload	1-3	1	3	3	4	1-2	2-3	2	4
Channel characteristics									
Cross-sections	1-3	1	4	3	4	2	2	1	4
Width/width-depth ratio	1-3	1	4	3	4	2	2	1	4
Pool parameters	1-3	1	4	3	4	1-2	2	2	4
Thalweg profile	1-3	1	4	3	4	1-2	2	2	4
Habitat units	2	1	4	4	4	1-2	2	2	4
Bed material size	2	1	4	3	4	1-2	2	2	4
Embeddedness	1-3	1	4	3	4	1-2	2	2	4
Surface vs. subsurface	2	1	4	3	4	1-2	2	2	4
Large woody debris	1	4	4	3	3	1	1	4	4
Bank stability	1-3	1	3	2	4	1-2	2	1	1-3
Riparian									
Canopy opening	1-3	1-2	3	1	4	1-3	1-3	2	3
Vegetation	1-3	3	3	1	4	1-3	1-3	1	3
Aquatic organisms									
Bacteria	4	4	4	4	4	4	4	1	1
Algae	1	2-3	2	2	3-4	1	1	1	3
Invertebrates	1	1	3	3	2	1	1	1	3
Fish	2	1	3	3	2-3	2	1	1-2	1-3

[1/] Mechanical and other methods that expose more than 30 percent of the soil.

[2/] Mechanical and other methods that expose less than 30 percent of the soil.

Quality Assurance/Quality Control

Quality assurance/quality control (QA/QC) planning and implementation are vital parts of any monitoring project. The QA/QC program will vary by level of effectiveness monitoring and the variable(s) being monitored. The following are some basic ingredients of a QA/QC program (EPA 1992).

Project Description: The specific monitoring goals and objectives should be explicitly described in the project description. The project description should describe how the data will be collected to meet the goals and objectives.

Project Organization and Responsibility: The organization of a monitoring project defines how the project will be conducted. A description of organizations and their functions should strengthen monitoring performance and compliance with QA/QC procedures. The key individuals responsible for collecting data and for ensuring precision and accuracy of data analyses should be identified by title, level of expertise, and a brief description of their responsibilities. An individual's level of expertise is especially important when an activity relies on professional judgment.

Quality Assurance Objectives for Measurement Data: For each variable measurement or value, the following QA objectives should be presented.

a. *Precision*: Precision is a measure of mutual agreement among individual measurements or values of a variable taken under similar conditions.

b. *Accuracy*: Accuracy is the degree of agreement between a measured value and the true or expected value for the variable.

c. *Completeness*: Completeness is the percentage of measurements made that are judged to be valid.

d. *Representativeness*: Representativeness is the degree to which data accurately and precisely represent a characteristic of a population, variations at a sampling point, or an environmental condition. The program should be designed so that the samples collected are as representative as possible of the habitat or population and a sufficient number of samples is collected.

e. *Comparability*: Comparability is a measure of the confidence with which one data set can be compared to another. Comparability is not quantifiable. However, it must be considered when designing sampling plans, analysis procedures, quality control, and data reporting. Employing consistent data forms and survey protocols will maximize comparability.

Laboratory Analytical Procedures: Methods of sample and data analysis should be specified. For EPA-approved or standard methods, pertinent literature should be referenced. For nonstandard or modified methods, detailed operating procedures should be provided, including sample preparation and analytical procedures.

Field Sampling Procedures: Field sampling procedures should include steps to be taken to ensure the quality of samples and sample data. The sampling procedures will vary by variable. The following are examples of procedures to documented:

a. Specific physical, chemical, biological, and habitat variables to sample
b. For biological variables, identify target assemblages
c. Sampling methodology
d. Habitat-assessment methodology
e. Details of sample preservation
f. Use and calibration of instruments
g. Replication and QC requirements
h. Sampling site selection

Sample Custody: A chain-of-custody procedure should be developed to ensure a written record traces the possession of the sample from the time of collection through data analysis.

Calibration Procedures and Frequency: A program of calibration procedures should be written to ensure that field and laboratory equipment is functioning at an optimal level.

Preventive Maintenance: A plan for routine inspection and preventive maintenance should be developed for all field and laboratory equipment and facilities to ensure data of consistently high quality.

Data Reduction, Validation, and Reporting: This part of the QA plan is designed to ensure good data by maintaining data quality throughout data reduction, transfer, storage, retrieval, and reporting. For example, biotic samples should be checked for proper taxonomic identification and forms checked for completeness, recording errors, plausibility, and consistency.

Internal Quality Control Checks: All personnel participating in monitoring activities must be trained in the proper use and maintenance of sampling equipment. They must be aware of the limitations of each piece of equipment. Internal quality-control checks can include replicate samples at stations to check consistency of collection, repeat field collections by separate crews, etc.

Data Precision and Accuracy Procedures: The best approach to ensure precision and accuracy is ensure that project personnel have received adequate training and gained the necessary experience to conduct the project. Adequate training of personnel in methods application is best way to ensure consistency, repeatability, and precision.

Performance Audits, Systems Audits: Quality control checks on the procedures used by field personnel should be done periodically during the field season. If any problems are found, corrective action must be taken immediately.

Corrective Action: A corrective-action program capable of detecting errors at any point in the project implementation process should be developed. The program must be able to identify problems and their source, implement action to correct them, document results of corrective action, and continue the process until each problem is eliminated.

Quality Assurance Reports: A formal report should be written to inform appropriate managers about the performance and progress of the monitoring plan.

Statistical Considerations

In designing a water-quality monitoring effort, statistical design is vitally important in drawing sound conclusions from the data. Statistical design will guide the sampling, the number of sites to sample, the statistical analyses to use, and the confidence level of the results. The following statistical discussion, excerpted from MacDonald et al. (1991), is lengthy but is necessary to demonstrate the relevance of statistics to water-quality monitoring.

Statistics are an inherent component of nearly all water quality monitoring programs. In most case a precise formulation of the monitoring objectives results in a question that is best answered on a statistical basis. For example, a common objective of water quality monitoring plans is to determine if a particular management activity is causing an adverse change in water quality. To answer this question in a quantitative manner, it is necessary to acquire data and make a comparison to other site(s), or to data from the same site prior to the management activity. If the monitoring plan is properly designed and replicated, data analysis will yield specific conclusions with an identified level of risk.

Statistics provides the scientific basis and procedures for studying numerical data and making inferences about a population based on a sample of the population (Mendenhall, 1971; Sokal and Rohlf, 1981).

By its very nature, water quality monitoring is a sampling procedure. It simply is not possible to make continuous measurements of all parameters at all locations. This means that before any data are collected one must address questions such as:

- *How many samples are likely to be needed to characterize a parameter with a specified degree of uncertainty?*
- *How many samples are likely to be needed to determine if there is a difference between locations, or a change over time?*
- *Where and when should samples be taken?*
- *Which parameters should be measured?*
- *How will the precision and accuracy of the data be assured?*

As the monitoring plan develops and data are collected, there is a continuing need to analyze the data, evaluate whether the data are meeting the objectives, and determine whether the timing and location of sampling is optimal. All these aspects of a monitoring program either require or involve statistics.

Many people react negatively to the use of statistics. Typically this is due to a lack of understanding about the role of statistics in water quality monitoring, or past experiences in which the application of statistics led to unexpected conflict or uncertainty. Statistics can make a strong positive contribution to water quality monitoring by:

- *providing an overall design for collecting and analyzing data;*
- *facilitating the precise specification of objectives, including an explicit recognition of the uncertainty and potential errors;*
- *providing a quantitative means to optimize the location and times of sampling, and thereby reduce costs;*

- *providing a rigorous set of procedures for analyzing the data collected in a water quality monitoring program; and*
- *providing a quantitative basis for making inferences about the characteristics and response of the population being sampled.*

To take full advantage of these potential benefits, those responsible for preparing monitoring plans should consult with a statistician early and often. Too often a statistician is consulted after the data have been collected, and the statistician's tools are unable to salvage inconsistent or unreplicated data.

These five contributions imply that statistical procedures are critical tools in water quality monitoring, but they are not a substitute for decision-making. Averett (1979) states "data interpretation is an intellectual activity; statistical applications is a mechanical activity." Those responsible for a water quality monitoring program still must decide how much uncertainty can be tolerated and balanced the relative risks and costs associated with different types of errors. The manager and technical staff must also determine what type of monitoring design is most appropriate, which parameters to measure, and the initial times and locations for sampling.

This chapter presents some of the key statistical principles which must be considered in developing a water quality monitoring program. The overall goal is to demystify the role of statistics and statisticians in water quality monitoring programs. The specific objectives are to: (1) explain how statistical considerations should be taken into account in designing and implementing water quality monitoring programs; and (2) explicitly discuss the trade-offs between sample size, inherent variability, level of significance, statistical power, and the minimum detectable effect.

Specific guidance on the selection and use of statistical test is not addressed in this document.

Statistical Design in Water Quality Monitoring

General Design and Replication

The overall design of a monitoring project is largely determined by the monitoring objectives and closely tied to the type of monitoring. In many cases the design of the monitoring plan will determine the statistical procedures used to analyze data.

Standard statistical designs are based upon a series of experimental units. Experimental units are defined as the objects upon which measurements are made (Mendenhall, 1971), and in water quality monitoring these are usually sample sites. In an idealized, simple experiment, the experimental units would be randomly selected and half assigned to some treatment, while the other half would be left as untreated controls. Both the treated and the untreated experimental units usually are considered to be representative samples of much larger populations. Repeated measurements on the experimental units generate the data used to describe the sampled populations,

and to draw inferences about the larger population from which the experimental units were drawn.

Multi-factor designs allow this simple experimental design to be expanded to multiple levels of treatments and their interactions. Multivariate statistics are used to analyze data and test hypotheses when several independent (causal) or dependent (response) factors exist. Multi-factor designs and multivariate statistics will not be discussed here as they are based on the same principles as the simpler, univariate methods.

Nonparametric statistics originally were developed to analyze qualitative or ranked data, and they also can be used when the underlying distribution of the data is not normal. Hence they are more broadly applicable and more robust in terms of requiring fewer assumptions, but they generally are less sensitive. Recent advances are likely to greatly increase the use of nonparametric procedures. In this chapter parametric statistics are emphasized because most water quality data are numerical and can be transformed into an approximately normal distribution.

The idealized simple experiment outlined above illustrates several key elements common to all statistical designs. First, a population is defined, and samples are drawn from that population. The population might be defined as a particular fish species in a stream reach, in pools of a certain size, or in a certain type of stream. Second, some treatment is applied to the designate experimental units, and this might be timber harvest, forest fertilization, or gravel extraction. Third, this treatment is applied to two or more experimental units, and two or more experimental units are left as controls. Fourth, a series of measurements are made, and these provide the raw data for the statistical analysis.

Unfortunately most water quality monitoring plans do not fit this idealized design. A typical objective of water quality monitoring plans is to determine whether the value of a parameter has changed over time at a particular site. The two most common approaches used to address this question are (1) to measure the selected parameter over time at the site of interest, as in trend monitoring; and (2) to compare data from a treated site with an untreated site, as in project monitoring.

With regard to the first case, there is only one experimental unit, and the data collected are a sample of all possible measurements in time. In some cases the onset of a management activity can be used to separate the data into two groups (i.e., before and after), and one can test for significant change over time by comparing the means and variances over the initial period (baseline data) to the means and variances following the onset of the management activity. Often, however, this straightforward approach is not valid because the data are serially correlated (i.e., the value of any given data point is related to the previous value), or the data vary according to season, discharge, or other variables.

The approach to detecting trends will depend on the number of data points available and the type of trends or correlations present in the data. Graphing the data is the first and probably most important step in identifying the complicating factors and determining the appropriate statistical approach (Gilbert, 1987). A basic choice is either to attempt to remove the trend or correlation and then use parametric statistics, or use nonparametric statistics on the original data. Gilbert (1987) provides a useful guide to trend analysis techniques, and he references Harnes et al. (1981) for analyzing discharge-related parameters and Montgomery and Reckhow (1984) for analyzing serially correlated data. . . .

The first and most common design to evaluate changes over time is to monitor a single site. This approach is useful to detect seasonal or other trends, but a basic problem is that statistical inferences cannot be made either about the cause of and observed change at the monitoring site or about the cause of similar changes observed at other sites. Data from other sites are necessary for making inferences about other locations (Hurlbert, 1984).

The paired-site approach is the second design which often is used to evaluate change. In this design two sites are monitored, and a statistical relationship between the sites is established for the parameter(s) on interest. After this initial calibration period, one site is subjected to a treatment (e.g., timber harvest), and the other is left as a control. A significant change in the statistical relationship between sites is used to indicate a treatment effect. This is the basic concept behind paired-watershed experiments (e.g., Bosch and Hewlett, 1982).

The advantage of the paired-site approach is that the untreated or control site provides a basis for separating the treatment effect from other extraneous factors (e.g., climatic events). Nevertheless, this design still shares the same major flaw as the single-site approach, namely the lack of replication. In the absence of replicated treated and control sites, there is no information on the spatial variability of the parameters being measured. An estimate of the variability is necessary to make any statistically based inference about the cause of observed difference between the treated and control sites. Since in most case sites are not replicate, claims of cause and effect must be based on other information and not statistical testing (Hurlbert, 1984).

Multiple pairs of treated and control sites, although costly, usually result in a high sensitivity to change. Both the control and treated sites are subjected to the same extraneous factors, so the exclusion of these factors greatly increases the likelihood of detecting a treatment effect.

The paired-site approach is commonly used in project monitoring. Typically water quality is measured upstream and downstream of a particular activity, and the observed differences between sites are presumed to be due to the particular project or activity. However, the known differences in water quality and stream characteristics between upstream and downstream locations (e.g., Hainman et al., 1991) means that a pre-project calibration period is essential for unreplicated sites. As in the paired-site approach, the absence

of multiple treated (downstream) and control (upstream) sites means that the inference of cause-and-effect must be based on qualitative evaluation rather than statistical testing.

As suggested above, neither the single-site nor the paired-site approach fit into the traditional randomized designs described in statistical texts. In most water quality monitoring plans, the experimental units are streams, lakes, or sampling sites, and these cannot be randomly allocated among treatments such as clearcutting or road building. Typically the experimental units and treatment(s) are already specified, and the objective of the monitoring program is to determine if change has occurred. Sampling sites are often fixed by the presence of a bridge or other structure from which samples can be safely taken at high flows, or by assess to the drainage network.

The randomized block design may be the most relevant to water quality monitoring. Each block includes all of the treatments as well as a control. Treatments are randomly assigned to the experimental units within a block. Analysis of variance procedures are used to evaluate the differences between treatments in one or more blocks, regardless of variation among blocks. Thus the primary advantage of this design is to exclude extraneous factors (such as site differences, which occur between blocks) and focus on the differences between treatments within blocks. This makes the design statistically more robust (i.e., the results are reliable over a wider range of conditions).

Paired watersheds and upstream-downstream comparisons represent two of the simplest forms of a block design. The combination of a treated watershed and a control watershed form one, unreplicated block. Additional paired watersheds undergoing identical treatments result in additional blocks. To the extent that treatments are randomly assigned to each experimental watershed within a block, this yields a randomized block design, and statistical inference can be made regarding (1) the cause of any observed differences, and (2) the likely result of a similar treatment on other unmonitored sites that are part of the same population.

In the case of upstream-downstream comparisons, the upstream site usually acts as the control, and the downstream site serves as the treated site. Again the addition of paired upstream-downstream sites generates additional blocks. The problem with this design is that the treatments are not randomly assigned within each block, but are set according to a pre-determined and recognized site difference. The statistical design to resolve this problem is to replicate pairs without any treatment or project, and compare the upstream-downstream differences between these untreated pairs to the differences for the pairs where there actually is management activity. Alternatively, a relationship between each upstream and downstream location could be established during a calibration period, and a change in this relationship following management activities would indicate a treatment effect rather than a site difference.

In practice it is often assumed that natural variability overwhelms a consistent site effect between the upstream and downstream locations. In this case, a calibration period might not be necessary, and any difference between the sites should be due solely to the treatment being studied. Such an approach is inconsistent with basic statistical principles as indicated above.

The problem of separating site differences from treatment differences is particularly acute for many of the channel parameters... Channel cross-sections, pool parameters, and bed material particle size are all sensitive to environmental factors such as local geology and landforms, and they may exhibit considerable variation over relatively short distances. ...the problem of spatial variation can at least be alleviated by carefully identifying the sites to be monitored, and monitoring prior to initiating management activities. This available of pre-project data is critical to inferring management effects, and also influences the choice of statistical test(s).

As indicated earlier, more complicated designs have been developed to analyze multiple factors and the interactions between them. The primary problem associated with these designs in that the number of blocks usually is a product of the number of different levels for each factor, and the desire to examine several factors at once rapidly leads to a large number of experimental units. For example, an evaluation of the effects of two levels of nitrogen and two levels of phosphorous in streams with three repetitions of each combination and a set of controls requires a total of 15 experimental units. Adding in a three-level, qualitative factor such as the type of riparian vegetation (e.g., coniferous, deciduous, or no tree cover) increases the number of experimental units to 45. The use of these more complex designs will depend on factors such as the availability of experimental units, the estimated importance of the interactions, the monitoring budget, and the statistical trade-offs.

Benefits of Proper Statistical Design

The importance of the statistical design can be illustrated by an example. Suppose a land manager needed to determine if clearcutting increases the number of landslides. The staff reviewed a recent set of aerial photos and determined that there were 25 landslides on 5,000 acres of clearcuts, and 50 landslides on 20,000 acres of mature forest. Although the number of landslides per unit area proportionally was twice as high on clearcuts as mature forest, it is not possible to make any generalizations on statistical conclusions because data on the variability of the number of landslides on clearcut and forested areas was unavailable. Statistical analyses require multiple measurements in time or space, but the results cited above are from a single survey of one control and one treated experimental unit.

One might argue that statistical rigor may not be important in cases where the difference is relatively large, but what is the certainty associated with the statement that clearcutting increases landslides if there were 70, 80, or 90 landslides in the uncut forest? The overall trend in land management is towards increasing regulation to eliminate the obvious adverse impacts.

Hence increasingly sensitive techniques will be required to evaluate future management effects, and this will require proper statistical designs.

Collection of essentially the same data in the context of an overall statistical design can greatly increase the value of the data. In the above example, instead of treating the entire area as one experimental unit with two treatments (clearcut and forested), the staff should have identified the population of clearcut units and potentially harvestable forested units. Then data on the area and number of landslides in each clearcut and each forested unit should have been collected. This procedure would yield a data set consisting of the number of landslides per unit area of each potentially harvestable forested unit and each clearcut unit. Since these data represent a sample of a much larger population of clearcut and potentially harvestable forested areas, statistical techniques can be used to determine the following:

- *the mean and variance of the number of landslides in each land use type;*
- *the approximate shape of the underlying distribution of the data (e.g., normal, binomial, Poisson, lognormal);*
- *the significance probability associated with the observed difference in the number of landslides between the two land use types; and*
- *the likelihood of obtaining a false conclusion.*

A proper statistical design can greatly improve the efficiency of data collection. For example, if the staff thought that the difference in the number of landslides in the clearcut and forested units was relatively large, measurement might only be made on a random sample of 10% of the units. If this sample indicated a statistical significant difference, measuring the remaining units might be unnecessary, and a substantial saving in the cost of the study would be realized.

The potential benefits of an adequate statistical design are even more apparent if there are several sources of variation. In the above example, the frequency of landslides might be strongly influenced by slope steepness or the type of bedrock. If the sample size is sufficiently large, statistical procedures can be used to separate these factors. Such information is extremely useful for developing practical management procedures, such as identifying high-risk areas or predicting sediment to streams from landslides.

In many cases the critical factors are known prior to initiating the monitoring project. This a priori information can be used to construct strata to improve the sampling efficiency and the sensitivity of the statistical tests. The basic principle is that the strata should remove as much of the within-stratum variability as possible, thereby allowing the treatment effect to stand out from the "noise" of the data. Continuing with the previous example, the forested and clearcut units might be stratified by geologic type. A random sample of the forested and the clearcut units would be taken from each geologic type (stratum), and an analysis of variance procedure would be used to detect differences in landslide frequency among strata (e.g., sandstone or shale) and treatment (e.g., clearcut or forested). If prior information is lacking, a pilot study can be extremely helpful in determining an appropriate statistical design, identifying strata, and estimating sample size.

Design Problems and Constraints

Some of the major problems and constraints associated with developing water quality monitoring plans include:

- *lack of adequate information prior to initiating a monitoring project,*
- *difficulty in distinguishing between the effects of management activities and natural events,*
- *difficulty in distinguishing among the relative effects of multiple management activities,*
- *the possible time lag between an action and its effect on water quality, and*
- *the random nature of climatic events.*

The lack of adequate information about the parameters to be monitored is an important limitation to the development of a monitoring plan. In many cases key sampling decisions must be made with little or no data on the diurnal and seasonal fluctuations of different parameters, the dependence of the parameters on flow and site conditions, and the spatial variability. This is why it is so important to consider monitoring as an iterative process. As monitoring proceeds more of the temporal and spatial variability is captured in the data set, and the quality of the information increases. This may make the initial design inappropriate, and the monitoring plan should be revised. Unfortunately any change in monitoring sites or methodology may preclude comparisons with the earlier data, thus making it important to conduct a pilot monitoring project before initiating large-scale or long-term monitoring activities. Even if the overall design is known to be satisfactory, a regular review of the monitoring objectives and data is needed to maximize the efficiency of data collection and analysis.

One example of how inadequate information might seriously hinder monitoring efforts may be found in the use of habitat types. At present almost no information is available on changes in habitat units over time, particularly in response to management activities. Similarly, few data have been published on the accuracy of habitat unit surveys under different flow conditions or by different survey teams. Nevertheless, extensive efforts are underway to characterize and monitor habitat types in streams, and arbitrary limits on the amount of allowable change are being established. A few years' experience with a series of pilot projects might be preferable to determine the spatial and temporal variability, help identify allowable limits for change, and determine whether a multi-stage sampling scheme could reduce survey costs.

A second constraint on the use of standard statistical procedures in water quality monitoring is the need to separate the effects of management activities from natural events. This is best done by comparing changes in water quality at unmanaged controls sites to changes at sites with management activities. The unmanaged control sites provide an index of change due to some key factors, such as climate, and the removal of these factors increase the sensitivity of the monitoring. Problems with this approach include the additional cost of monitoring the control sites, and the fact that in many areas it is becoming increasingly difficult to find sites that have not been subjected

to extensive management activities. Valid control sites also must be left undisturbed for the duration of the monitoring program. Although it may be possible to find adequate control sites for small headwater streams in some areas, most of the control sites for larger streams will be found only in parks or wilderness areas. This suggests an increasing distance between the treated and the control sites, and an increasingly tenuous assumption of comparability. With an weaker statistical relationship between sites, there is a declining ability to detect significant changes due to management activities.

To a certain extent additional observations can substitute for a rigid statistical design. Qualitative or quantitative data that document the processes linking management activities to water quality can greatly enhance the validity of any observed trends in water quality. For example, direct observations during storm events can show how a particular road or clearcut is affecting water quality, and this can help compensate for the absence of replicated control sites.

A third basic limitation to using standard statistical procedures is the problem of overlapping activities. Relatively few watersheds are subject to just one management activity. Often one type of management activity, such as forest harvest, incurs other activities, such as increased road traffic, additional road maintenance, and road construction. Larger watersheds, which are the focus of most trend monitoring programs, tend to have more numerous and diverse land uses. Since in-channel water quality measurements integrate the effects of all the upstream management activities and a myriad of natural processes, in many cases a change in water quality cannot be directly related to a specific management activity or a specific set of BMP's. This may not be important if the monitoring objective is simply to determine the overall condition and trend in water quality. However, the common objectives of most water quality monitoring programs are to identify problems, minimize adverse impacts, and guide future management. The limited capacity of inchannel measurements to distinguish among overlapping management activities means that these objectives may not be met through inchannel measurements. Multivariate techniques offer the potential to resolve the effects of several independent (causal) variables, but their use in water quality monitoring is hampered by the large number of variables needing consideration, and the limited number of monitoring sites that can provide the necessary dependent (effect) data.

The fourth limitation to using standard statistical procedures is the potential lag time between a management activity and its effect on water quality. For example, several studies have shown that the hazard of landslides in clearcut areas is maximized 4-15 years after the harvest is completed (e.g., Gray and Megahan, 1981; Swanston, 1969). Similarly, the time lag between the detachment of a soil particle and its delivery to the stream channel is often substantial (Swanston et al., 1982). This suggests that the typical project monitoring period of up to 3 years (Ponce, 1980a) may not be adequate to evaluate the long-term or delayed effects of certain management activities on water quality.

Climatic variability poses a similar problem with regard to the design of water quality monitoring projects. The relative impact of a particular management activity may vary according to the severity of the climatic conditions during the period of maximum management impact. This suggests that a longer monitoring period may be necessary if one wishes to evaluate or contrast management impacts with the more sporadic or extreme natural events. In some cases extreme events, such as floods or debris flows, can completely overwhelm the changes in water quality due to management activities (e.g., Lisle, 1982; Griggs, 1988). Statistics can be used to evaluate the likelihood of experiencing a certain event within a specified time period, and this information can be helpful in the initial formulation of a monitoring plan. Similarly, the results of a water quality monitoring project must be evaluated in the context of the climatic events experienced during the monitoring period.

For some water quality monitoring objectives, a statistical design may not be necessary. A source-search methodology often can be used to qualitatively identify the cause of a water quality problem, or the most likely locations for sampling. The basic procedure is to make systematic observations, usually in the upstream direction, in order to identify the source of potential or existing water quality problems. Often observations or water samples are taken at each major tributary. This procedure is most effective if there is a localized pollution source that is having a substantial impact on water quality, and when measurements can be made in the field.

Similar procedures can be followed to rapidly and qualitatively evaluate management practices and impacts. Walking or driving a road network during runoff events, for example, can provide a useful, qualitative review of BMP's, and indicate where road-related water quality problems are developing. This type of reconnaissance can be extremely cost-effective as it facilitates the early identification of problems without embarking on a costly monitoring scheme. Such activities also offer the potential to resolve adverse impacts at an early stage, and thereby reduce the costs of repairs or future mitigation measure.

In short, there is a need to complement any instream monitoring program with additional observations or measurements. These should aim at (1) providing a direct link between upslope or riparian management activities and instream water quality; and (2) enhancing one's understanding of watershed processes. Such information is not only helpful in alleviating any problems with the statistical design, but also is essential for helping to guide future research and management.

Principles of Sampling

Many of the most important principles of sampling are similar to the principles of statistical design, and these are discussed in most statistical texts. The three basic types of sampling are random, systematic, and stratified, and each can be applied in space or over time.

The procedure for simple random sampling is to clearly identify the universe of potential sampling times or locations, and then select individual times or

locations for sampling according to a random number table or any random procedure. If information on the variability of the parameter is known, then the number of samples needed to achieve a certain confidence interval can be calculated. For example, simple random sampling might be used to select the particular day for measuring pH in a large river.

Using simple random sampling to select monitoring sites may prove difficult in practice because it requires identifying all possible sampling sites (i.e., the sampling frame). This may not be a problem if the precise location of the sample is not important. The sites for monitoring many water column parameters, for example, could be randomly selected from the population of river miles. Simple random sampling could be very time-consuming if one wishes to sample only certain habitat types (Hankin and Reeves, 1988).

Systematic sampling consists of randomly selecting the first sample, and then selecting all subsequent samples by applying a constant interval. Systematic sampling can result in a biased sample if there is a systematic variation in the population being measured. For example, if the timing of the first sample in a given year was determined randomly, and subsequent samples were taken at exactly 6-month intervals, this might not represent a true long-term average because all the samples would be taken in two different seasons. Hankin and Reeves (1988) discuss the merits of different sampling schemes to estimate fish abundance and habitat areas in small streams. They advocate systematic sampling of individual habitat units (e.g., measuring the area of every tenth glide, or counting the fish in every fifth plunge pool) because it is the most practical and is unlikely to significantly bias the results. Systematic sampling along a river or stream can be an efficient means to detect distinct but unknown sources of pollution (Gilbert, 1987).

Stratified random sampling involves some grouping of the population of interest, and then randomly sampling each group or stratum (Ponce, 1980a). This procedure is often used in water quality sampling because certain parameters are known to vary by the time of day, season, discharge, or some other factor. The different strata can be sampled at different frequencies according to the estimated size of the population (proportional sampling) or the variability within the different strata (optimal sampling). Optimal sampling generally is preferable for flow-dependent parameters, whereas proportional sampling may be equally efficient for time-dependent (e.g., seasonally varying) parameters.

The advantages of stratified random sampling are similar to the advantages of a randomized block design, in that it can (1) improve the efficiency of sampling, (2) provide separate data on each stratum, and (3) enhance the sensitivity of statistical tests by separating the variability among strata from the variation within strata. The information needed to construct the strata and estimate the sampling frequency must either be known prior to sampling or obtained through a pilot study.

Seasonal strata are often used for sampling invertebrates and fish, while discharge is often used to establish strata for sampling sediment and the other physical and chemical constituents of water. Stratification by discharge also helps ensure that high flows are sufficiently sampled.

Sub-strata can also be defined. If discharge is used to define the primary strata, for example, the high flow stratum might then be sub-stratified according to the position of the hydrograph (e.g., rising limb or falling limb) or by the cause of the high discharge (e.g., snowmelt or rainstorm). The former might be appropriate in the case of turbidity or suspended sediment, while the latter might be more applicable for specific conductance, pH, or nutrient concentrations. Although the value of stratification can be statistically tested, a plot of the data provides a quick and qualitative indication of the value of stratifying. The statistical benefits of a reduction in unexplained variability must then be weighed against the cost of the sampling scheme needed to adequately characterize each stratum.

For water quality programs the decision regarding where to sample is largely determined by the objectives of the monitoring program, location of the management activities, layout of the catchment(s) to be monitored, access to the monitoring sites, and the design of the monitoring program. These issues are discussed in greater detail in Gilbert (1987), Kunkle et al. (1987), Ponce (1980a,b) and Sanders et al. (1987). For project monitoring the general principle is to locate the sampling sites as close to the actual project as practicable, as the largest water quality impact will be immediately downstream of the activity. Minimizing the distance between the upstream (control) and downstream (treated) sites will help minimize confounding site differences....

Empirical knowledge of the basin to be monitored is extremely helpful in developing a monitoring program (Ponce, 1980a). Even a cursory inspection can indicate the types of adverse change that are likely to be encountered and the spatial distribution of management activities. This type of spatial data provides much of the guidance needed to establish sites and direct the monitoring activities towards the problem areas.

Monitoring of channel characteristics or water quality within a particular basin or region may best be achieved with a spatial stratified sampling scheme. In keeping with the principles of stratified random sampling, the best approach is to classify stream segments and subsample these. Rosgen (1985) and Cupp (1989) are probably the two most widely used stream classifications systems at this time. Once the stream segments have been identified, an additional stratification into habitat units may be desirable (Hankin and Reeves, 1988). Specific details for laying out such nested sampling schemes are beyond the scope of this document, but the principles and references cited in this section can provide the necessary guidance.

In turbulent streams many of the physical and chemical constituents of water are relatively insensitive to the precise monitoring location. For these parameters a general site description, such as just upstream of a particular tributary, usually is sufficient. In less turbulent reaches, some parameters, such as suspended sediment, can vary considerably with depth, and the reader should refer to the appropriate U.S. Geological Survey publication for detailed sampling guidelines.

Other parameters, particularly those that pertain to channel geomorphology, may exhibit a great deal of variation over a few meters. For these parameters a more precise site description is needed, and this usually is based on prior knowledge. Bed material particle size, for example, might be evaluated in certain, geomorphologically determined locations like the downstream edge of a point bar. Embeddedness often is measured in riffles with certain characteristics, although the precise location of each hoop sample is random.

Even after determining the general time and location of sampling, another series of sampling questions must be addressed. Is a grab sample close to the bank adequate, or should a series of depth-integrated samples be taken across a particular cross-section? Is one sample in time adequate, or should several samples be taken? If several samples are taken across a channel or over a relatively short time, should these samples be kept separate or combined?

The answer to these questions largely depends on the objectives of the study, the parameter being measured, and the site characteristics. Certainly the same statistical principles apply to these fine-scale questions of sampling as they do to the larger-scale questions of location and timing discussed above. Composite samples over time or space can represent a substantial saving in analytic costs, but this reduces the resolution of the data. Samples for analyzing bacterial contamination should never be composited. If only a single grab sample can be taken, this should be taken in the middle of the stream at the 0.6 depth (Ponce, 1980a). Specific recommendations for sampling various parameters can be found in APHA (1989), Greeson et al. (1977), Guy (1970) and Guy and Norman (1970). Once a monitoring project has been initiated, any change in sampling procedure should be undertaken very cautiously, as this may preclude any comparisons to data collected using any other procedure.

Principles of Statistical Testing

Assumptions and Distributions

Most of the better-known statistical tests have been developed for scaler data that follow a bell-shaped or "normal" distribution. Data with these attributes normally are analyzed with parametric statistics. Nonparametric statistics are applied to data which have an unknown population distribution, or that are ranked or categorized rather than measured. For normally distributed data, nonparametric statistics are less efficient than parametric statistics (i.e., they are more likely to yield false conclusions) (Mendenhall, 1971). The lower efficiency of nonparametric statistics is a primary reason why transformations and other methods are used to obtain data that approximate a normal distribution. The advantage of nonparametric statistics for water quality monitoring is that they require fewer assumptions about the underlying distribution of data. The focus of the following section is on parametric statistics, but most of the principles also apply to nonparametric statistics.

The key assumptions for the use of parametric statistics are as follows:

1. *the data are normally distributed,*
2. *the data are a random sample of the population,*
3. *the observations are spatially and temporally independent, and*
4. *the errors in the data are randomly distributed.*

Each of these assumptions can be tested, but only major violations can be identified. Although these assumptions may not be strictly true in many cases, the relevant question is whether a violation of these assumptions substantially affects the probability statements being drawn from the data (Ponce, 1980b). Different statistical tests vary in their sensitivity to each of these assumptions. In uncertain situations nonparametric statistics can be used to bolster or supplement any conclusions developed from the use of parametric statistics. The following paragraphs briefly discuss these four assumptions for parametric statistics in the context of water quality monitoring.

1. *The first step in determining if the distribution of a data set is normal is to plot it. Data can be plotted over time, against a controlling variable such as discharge, or as frequency distribution. A plot of the raw data in important to visualize the distribution, as this allows a quick and qualitative check for patterns, extreme values, and obvious errors. Often a frequency distribution of water quality data shows a distinct clumping to the left with a long tail of extreme values to the right. This type of distribution is known as a lognormal distribution, and it usually can be converted to the normal, bell-shaped distribution by converting the data to base 10 or natural logarithms. If zero values are present, the data are transformed by adding 1 to each value and then taking the logarithm (Ponce, 1980b). After transformation the data should always be plotted again, as this provides a familiarity with the data and an intuitive check on the transformations and any subsequent calculations.*

   Numerous other transformations can be used, and two of the most common are the square root and cube root, respectively. Both transformations and normalization procedures are discussed in most statistics texts along with the tests necessary to check on the normality of the data. One "quick and dirty" test for the normality of a frequency distribution is to determine whether 2/3 of the data falls within on standard deviation of the mean, and 99.9% of the data is within three standard deviations of the mean. If this is the case, the data are likely to be considered normal for statistical testing purposes.

2. *The second key assumption for parametric statistics is that the data are a random sample of the population of interest...*

3. *The third assumption of spatial and temporal independence is best met by establishing a proper design for collecting data. Daily streamflow data, for example, are not independent in time, as the streamflow for any given day is partly dependent on the amount of flow in the previous*

day (i.e., they are serially correlated). One could, however, randomly sample from a population consisting of all the daily streamflow values. Similarly, there is often a strong correlation between data collected in adjacent basins, and this relationship forms the theoretical basis for the paired-watershed approach. Usually this problem is best addressed by properly defining the experimental units and the population to be sampled. For example, the rainfall and streamflow data for adjacent small basins will be highly correlated, but the rainfall for a particular day from a series of gages in comparable locations should be normally distributed. Thus an acceptable population for sampling might be the precipitation data for a particular day from a series of gages. However, statistical analysis of daily precipitation at a site would have to account for the autocorrelation between daily values, and this can be done through time series analysis (e.g., Box and Jenkins, 1976).

4. *The final assumption—the random distribution of errors in the data—also can be satisfied by ensuring that the design and sampling procedures do not contain systematic errors. This is done by randomly assigning treatments to the experimental units, by random sampling, and by careful attention to measurement procedures (Sokal and Rohlf, 1981). Systematic errors can be removed if the cause can be identified, and the removal of systematic errors is a key procedure in Hankin and Reeves' (1988) methodology for measuring habitat types. The danger is that systematic errors are not recognized, and these could easily result from a change in personnel, equipment, or measuring techniques. The possibility of systematic errors is a particularly important consideration when analyzing trend data from a single site. Quality assurance and quality assurance techniques (e.g., EPA, 1983) are an important means for reducing the possibility on non-random errors. Plots of the data and residuals can help identify unusual trends, and nonparametric tests can be used to assess whether the errors are significantly non-random.*

Statistical Compromises

Probably the most commonly asked question in water quality monitoring is whether significant change has occurred. The use of inferential statistics requires that this question be more properly phrased as "Can we conclude that there is a difference between population A and population B based on samples drawn from those two populations?" If the data are collected in the context of a proper statistical design and meet the criteria discussed in the previous section, then our ability to answer this question depends on five interacting factors:

1. *sample size,*
2. *variability,*
3. *level of significance,*
4. *power (i.e., the probability of detecting a difference when on exists), and*
5. *minimum detectable effect.*

The important aspect of these five factors is that a change in one factor will affect some or all of the other factors. In general, any improvement in one factor will come at some cost. This cost could be with regard to one or more of the other factors, increased sampling costs due to an increase in the sample size, or a change in the statistical design. The technical specialist and the manager must realize that there is no perfect solution in statistics, and an explicit recognition of these trade-offs is a necessary stage in designing a statistically-sound monitoring plan. The following paragraphs discuss each of these five factors and their relative effects on monitoring costs, uncertainty, and risk. Note that these trade-offs are discussed with regard to a comparison of the means from two populations using parametric statistics, but the same principles apply to all statistical tests using both parametric and nonparametric statistics.

Sample size. *The relationship between sample size, accuracy, and uncertainty is generally understood. For example, a larger sample size will reduce the difference between the sample mean and the true population mean. A larger sample size also will reduce the standard error of an estimated parameter. (The standard error is the standard deviation of a particular descriptive statistic, such as the sample mean. Usually it is estimated for a population from a single sample.) It follows that a larger sample size increases the ability to detect a difference between two populations because less uncertainty is associated with the estimated population means. Unfortunately this increased ability to detect a difference between two populations is a logarithmic function of the sample size rather than a linear function. This means that increasing the sample size may make a substantial difference if there are very few samples (e.g., less than five or ten), but the benefits of increasing the sample size beyond about thirty or forty generally are very small unless the parameter is highly variable.*

The statistical trade-off associated with a larger sample size is that it increase the likelihood of concluding there is a statistical difference when in fact there is no difference (Type I error; see discussion on power below). For most water quality monitoring programs Type I error is not a major problem. Typically the costs of sampling and the inherent variability mean that one is more likely not to detect a difference when in fact there is a significant difference (Type II error; see discussion on level of significance below). The other problem associated with increasing the sample size is that each additional sample has certain cost, and one must evaluate the marginal benefits of an additional sample as compared to its cost in terms of drawing resources away from other activities. If this marginal cost is acceptable, it usually is statistically advantageous to increase the sample size.

A variety of statistical techniques can be used to estimate the appropriate sample size give the statistical objectives and the known variability of the parameter. Often information on the variability of parameter can be obtained from previous studies. If no information is available, usually it is best to conduct a small pilot study. This will not only provide some basic information on the parameter(s) of interest, but also provide an opportunity to refine the

objectives and techniques. If no information is available and a pilot study is not possible, a sample size of "about 10" often represents a reasonable compromise between the cost of sampling and the need to reduce the uncertainty of the population estimates. This estimate is based on the minimum detectable effect for the t-test (Figure 3).

***Variability**. The most common statistic used to describe the variability of a sample is the standard deviation. The square of the standard deviation is the variance. Both the standard deviation and the variance are expressed in the same units as the mean. Dividing the standard deviation by the mean yields the coefficient of variation, and multiplying this by 100% provides a standardized measure of the relative variability in percent of the mean.*

As suggested above, the variability of a parameter in time and space is inversely related to the ability to detect significant change. Increasing the sample size can help compensate for a high degree of variability, but since the standard deviation is a square root function this is subject to diminishing returns. The other, more difficult means to decrease the variability is by improving measurement techniques and sampling methodology. Errors in measurement can spring from a wide variety of sources, and these include equipment problems, actual measurement problems, transcription problems, and inaccurate data entry. Reducing these errors can be very time-consuming, but it is a necessary part of the quality assurance and quality control aspects of any monitoring project.

The other effective means to reduce the variability of the data is to modify the sampling scheme. If some of the factors causing the variability can be identified, then the data can be more efficiently stratified. A series of statistical techniques can be applied to evaluate the efficiency of existing strata, and to optimize sampling among the different strata.

In some cases a reduction in variability can be gained only by narrowing the scope of the investigation or objectives. Often the natural variability in the streams being monitored makes it difficult to answer broad management questions. Narrowing the question allows a more focused investigation, a reduction in unaccounted variability, and improved statistical resolution. For example, bed material particle size might be evaluated only in a very specific location, such as the deepest portion of certain types of pools, or the downstream edge of point bars. A change in fish populations might be more narrowly defined as a change in the number of 1 + steelhead. Basins subject to landslides or debris flows might be considered separately. One also might reevaluate the parameters being measured. In contrast to simply increasing the sample size, all these approaches require more information about the streams and the parameters being monitored, and hence more involvement of the technical staff. This has advantages for interpreting the monitoring results, and for designing future monitoring plans.

***Level of significance**. The level of significance refers to the probability that an apparently significant difference is not real but simply due to chance. This*

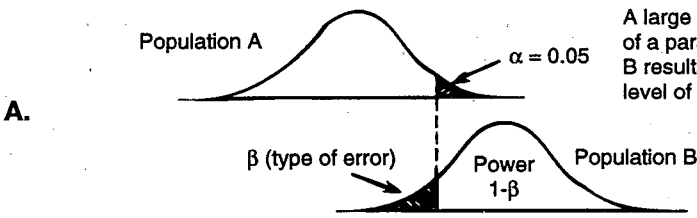
is the α value listed in statistical tables and shown graphically in Figure 2. Convention has the α value set at 0.05 for most statistical tests, and this means that there is only a 1 in 20 chance that an observed difference is due to chance (Type I error). A stronger level of significance (i.e., lower α) indicates a higher lever of confidence that the difference is real, and a lower probability that it was due to chance. The most common ways of obtaining as stronger level of significance are to: (1) increase the sample size, and (2) reduce the variability by altering the measurement techniques or sampling design.

Figure 2A graphically shows that a comparison of samples from two normally distributed populations with quite different means has both a strong level of significance as indicated by the shaded area in Population A where a = 0.05, and a high power (i.e., a high probability of detecting a difference when in fact there is a difference) as indicated by the large unshaded area in Population B. Figure 2B indicates that as the two populations become more similar, there is a decreasing ability (i.e., declining power) to detect a significant difference when the level of significance is kept at 0.05. Figure 2C is similar to Figure 2B, except that the variability of population A and B has been reduced (i.e., the estimated means of the two populations have less uncertainty). This increases the power for the same level of significance.

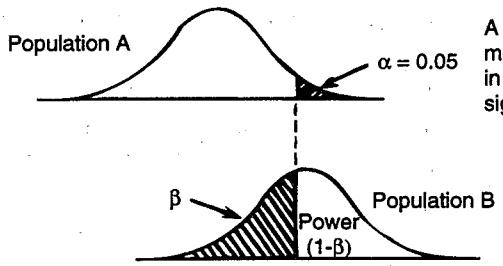
The selection of an α level is purely arbitrary and should reflect the values and risks associated with each of the other four factors discussed here. In most cases α is set at 0.05, but for many water quality applications a higher α level (or weaker level of significance) may be more appropriate. One justification for a higher α is that an objective of most monitoring programs is to identify changes in water quality due to management. However, by the time an adverse change has been detected, adverse effects on the designated uses may already occurred, and restoration or recovery may be a long or costly process. Hence it may be preferable to try and identify change earlier by decreasing the level of significance, even though this will simultaneously increase the likelihood of identifying change when it has not actually occurred. In other words, the cost of not identifying adverse changes is greater than the cost of erroneously detecting change, and this is the basis for the trade-off between the level of significance (α) and power ($1 - \beta$).

Figure 2D illustrates that a weaker level of significance will increase power. If there is considerable overlap between the two populations, a substantial increase in power can result from a relatively small increase (weakening) in the level of significance.

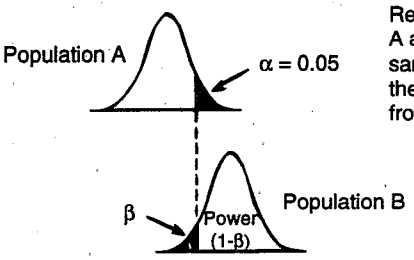
An example of a situation where a weaker level of significance might be appropriate in monitoring for bacterial contamination is a stream used for domestic water supplies (Ponce, 1980b). In this case the cost of failing to detect contamination is quite high, and it usually is better to have more false warnings that to miss a contamination episode. Other reasons for adopting a weaker level of significance include the high variability of water quality parameters, and the costs associated with increasing the sample size in order to achieve a stronger lever of significance.



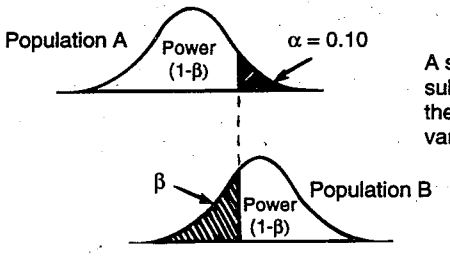
A. A large difference in the estimated mean values of a parameter for population A and population B results in both high power (1-β) and a strong level of significance. This is the ideal situation.



B. A smaller difference between the estimated mean values for populations A and B results in lower power at the same level of significance.



C. Reduction in the variability of populations A and B results in more power at the same level of significance, even though the population means are unchanged from Figure 5B.



D. A slightly weaker level of significance substantially increases the power for the same population means and variances as in Figure 5B.

Figure 2. Schematic representation of the trade-offs among level of significance, power, and variability for two normally distributed populations. The figures assume a one-tailed t-test is being used to determine if a significant difference exists between the two populations.

One should also keep in mind the distinction between the statistical level of significance and the level of significance relevant to the designated uses of water. A 25% decline in outmigrating salmonids may not be statistically significant because of the large interannual variability, but a loss of 25% of the outmigrating salmonids due to poor water quality or habitat deterioration is a serious impairment of the designated use for fisheries. Conversely, a level of significance of 0.05 for a given water quality parameter does not necessarily mean that a designated use is impaired. The point is that the statistical results must be interpreted by the specialist and the manager, and this requires an understanding of the physical and biological functioning of the water body being monitored.

Power. *The power of a statistical test is the probability of detecting a difference when in fact there is a difference (Mendenhall, 1971). This probability is usually designated as $1 - \beta$. When comparing two sample means, the quantity of β is commonly known as Type II error. Type II error can be described as the probability of incorrectly concluding that two populations are the same when in fact they are different. Since power is closely related to the level of significance, it exhibits a similar response. An increase in sample size usually will increase the power of a test by reducing the uncertainty around the mean. The greatest increase in power occurs at small sample sizes. More variability increases the overlap between two populations, thereby decreasing the power of a test. Decreasing the minimum detection limit will increase the allowable overlap between two populations. This enhances the possibility of Type II error and correspondingly reduces the power.*

Minimum detectable effect (MDE). *A key factor that should be considered in the design phase of most monitoring programs is the minimum change one wishes to detect. Usually this question is not explicitly considered, even though it is directly related to sample size, parameter variability, level of significance, and statistical power (Green, 1989). Any decision regarding the desired minimum detectable effect also must consider the sensitivity of the different designated uses in the streams being monitored. In other words, how much change is acceptable in the parameter being monitored before a designated use is impaired? Although in some cases the answer might be that no change is acceptable, this not a statistically acceptable answer because no monitoring program can detect an infinitesimal change. An explicit discussion of the MDE is helpful in forcing the manager and the technical staff to agree on specific, quantitative objectives for the monitoring plan and, by implication, for management impacts.*

For some parameters, such as pH, a change in either direction is significant, and this expands the allowable zone of change if the level of significance is kept constant (i.e., a two-tailed test instead of a one-tailed test). If the concern over possible change is only in one direction, such as an increase in suspended sediment concentration, then the limit of allowable change will be slightly less (i.e., a one-tailed test).

For most trend, project, and effectiveness monitoring, an explicit MDE target should be established. Setting an MDE in validation monitoring may be helpful in defining the uncertainty in the model being validated. The MDE may be an important issue in compliance monitoring it the numerical standard is expressed in terms of percent change above background. Monitoring for such a standard is often difficult because of the need to first determine background, and then to evaluate the numerical change in a sometimes highly variable parameter. For example, some States permit forest harvest activities to increase turbidity by only 10 or 20% above background. In view of the temporal variability associated with turbidity, this standard becomes very difficult to enforce.

Figure 3 graphically shows the percent change detectable according to sample size and the coefficient of variation. This assumes a constant significance level of 0.05 and a power of 80%. Changes in the significance level (e.g., to 0.10) or power (e.g., to 90%) would not substantially alter the form or values in Figure 3. In very rough terms, Figure 3 indicates that the treatment effect, expressed as a percent of the mean, will have to be somewhat larger than the coefficient of variation for detection when $\alpha = 0.05$. Increasing the sample size is an increasingly effective tactic to decrease the MDE as the coefficient of variation increases. If a relatively small MDE is desired, one must have a correspondingly low coefficient of variation, and this has important implications for selecting parameters.

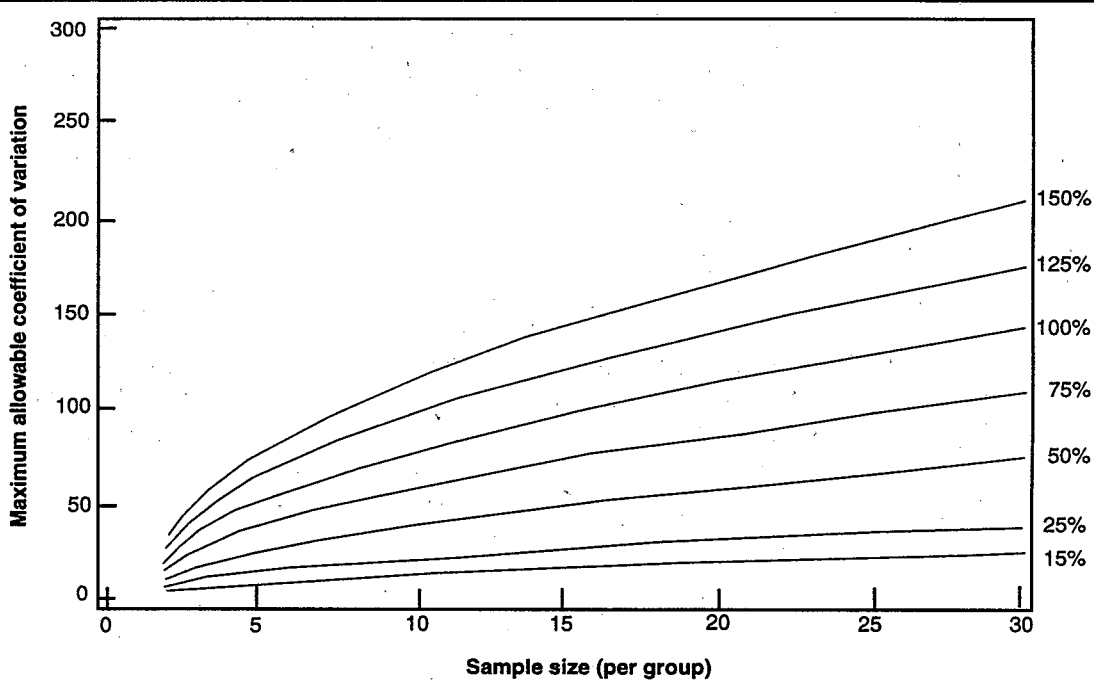


Figure 3. Maximum allowable coefficient of variation to detect changes ranging from 15 to 150%. Figure assumes a two-sample t-test is being used to detect change at a 5% level of significance and a power of 80%. The labeled curves show the minimum percent change that can be detected given a particular coefficient of variation for the parameter being measured and population sample size (figure courtesy of L. Conquest, Center of Quantitative Studies, University of Washington).

Selecting the Appropriate BMP Effectiveness Monitoring Level

Monitoring Levels

The monitoring level selected must be appropriate to the objective of the monitoring program. Four levels of monitoring are used for evaluating the effectiveness of forestry BMP's in meeting water-quality goals or standards: BMP I, II, III and IV (Table 3). They are defined as follows:

BMP Effectiveness I uses empirical observations and a limited amount of qualitative data to evaluate effectiveness. A heavy reliance on experience is used to judge whether land management or BMP's are effective in protecting water quality. It is a professional judgment as to whether there is an obvious problem or not.

BMP Effectiveness II depends mainly on qualitative data and limited quantitative data for analysis leading to the evaluation of effectiveness.

BMP Effectiveness III relies predominantly on quantitative data, with some qualitative data for detailed analysis to evaluate effectiveness.

BMP Effectiveness IV relies predominantly on quantitative data, with some qualitative data for detailed analysis to evaluate effectiveness, to establish water-quality criteria, goals, or cause and effect. This level of monitoring is usually associated with research.

Low-level monitoring (BMP Effectiveness I and II) identifies high-level needs and also extends the information developed by high-level monitoring. High-level monitoring (BMP Effectiveness III and IV) develops information that supports or validates decisions made in low-level monitoring.

The level selected is determined by:

1. The type of question or issue to be addressed.
2. The type, number, and skill of people required.
3. The number and stratification of reference and study streams.
4. Representative monitoring methods available for evaluating: (a) on-slope conditions, (b) chemical and physical components, (c) channel geomorphology and stability, (d) biology, (e) habitat, and (f) aquatic vegetation (Table 5).
5. Amount of data to be collected, time required, and number of streams that can be monitored,
6. The quality of data needed for making decisions.
7. The quality of decisions needed and the potential risk in making the wrong judgments.

Tables 3 and 5 should be viewed as guides only, not to be rigidly followed in selecting monitoring methods to use. For example, a user can use a level 3 method at level 2 to answer a specific question.

Table 3. Effectiveness monioring process matrix.

Monitoring Level	Questions or Issues	Skill Levels	Reference and Study Streams Used	Amount of Data Collected	Number of Streams Evaluated	Time to Decision	Quality of Data For Decision	Quality of Decisions and Risk
BMP I	Complaint Screening BMP effectiveness for obvious yes or no. Early warning on BMP effectiveness.	One or 2 trained professionals w/ technician.	State reference stream for ecoregion or establish a reference comparable to study stream. above and below.	Small to moderate amount.	Numerous	A few hours to 1 or 2 days to a week.	Qualitative data and empirical observation; limited data. Obvious good or bad recognized with large grey area.	Decisions are subjective, with high risk of making wrong decision for grey area situations.
BMP II	Complaint-high value stream. BMP Effectiveness	2 professionals trained in hydrology, fisheries, habitat, and invertebrates, + technicians.	2-3 Reference & 2-3 study streams, plus stream order, channel class, riparian and slope vegetation.	Moderate to large amount.	Many	2 weeks to a month.	Qualitative and quantitative data; moderate amount of data and precision: moderate grey area.	Decisions are more objective, yet judgement plays a major role in the decision. The grey area is reduced,thus reducing risk.
BMP III	Complaint-high value & public interest stream. Desired future condition. Compliance with water quality standards. Modifying BMPs.	Professionals in statistics, hydrology, fisheries, invertebrates, channel geomor-phologists plus technicians.	3 or more reference & study streams, depending on statistical design, using detailed stream/ecosystem stratification.	Large to very large amount.	Limited number.	One time visit to the refer - ence/study streams--2 to 3 months. Repeated visits for some issues will require 1 to 3 years.	Quantitative data with limited qualitative data. Good precision and a large volume of data. Good precision. Able to detect most signif icant impacts and grey area is small.	Objective and informed decisions based upon data. The risk is low. The decisions are defensible.
BMP IV	Cause & effect. Modifying BMPs. Law suites.	Same as level 3, but many are likely to be researchers.	Same as level 3.	Very large amount.	Very limited number.	2, 3, or more years.	Quantitative data with very little qualitative data. Good precision, able to detect small changes. Very small grey area.	Objective, data based decisions, which are very defensible. The risk is low.

There are two ways to interpret Table 3—horizontally or vertically. The horizontal presentation shows the various factors involved in the monitoring process. The vertical approach documents the differences among levels of monitoring and shows the progression from low to higher levels of monitoring, allowing for rational discussion of monitoring subjects. The vertical approach is also used for discussions of representative monitoring methods (Chapter 6).

Monitoring Objectives

The type of question or issue raised will define monitoring objectives. These objectives, and the associated significance to society and to the manager, will determine the monitoring level to use. Certain kinds of questions or issues call for specific monitoring levels. The following list of questions or issues is not exhaustive, but suggests the level or nature of the objective(s) suited to a specific level of monitoring.

BMP Effectiveness I

A. Issue: A complaint about a management practice's impact on a stream. This monitoring response would quickly determine if there is an obvious impact. If effectiveness is not empirically obvious, a higher level of monitoring might be needed to determine whether management is protecting the physical, chemical, and biological integrity of streams.

B. Issue: Need a quick and inexpensive way to determine which BMP's are obviously meeting water-quality goals.

C. Issue: Need early warning as to which BMP's are effective.

Caution: *Low levels of monitoring will detect which management activities, with proper implementation of BMP's, obviously have or have not protected water quality, but uncertainty will prevail in many cases. These will require more detailed data collection and analysis to determine effectiveness and could call for a higher level of monitoring. Careless or poorly trained observers will tend to classify BMP's as effective when in fact they may not be.*

The key to accurate professional judgment is experience and knowledge gained from monitoring streams in the ecoregion at higher levels and applying that experience and knowledge at this level in the same ecoregion. An alternative is sound training by individuals experienced in the ecoregion.

BMP Effectiveness II

A. Issue: Need to respond to a complaint concerning a high-value stream.

B. Issue: Need to respond to a concern by a water-quality agency.

C. Issue: Trends of water quality.

BMP Effectiveness III

A. Issue: Need to respond to a complaint about management impacts on a high-value stream with high public interest, such as outstanding national resource waters or water-quality-limited streams.

B. Issue: Need to define biological criteria for water-quality standards.

C. Issue: Need to define desired future conditions under Forest Land Management Plans.

D. Issue: Forestry compliance with State water-quality standards.

E. Issue: Determining the need to modify BMP's.

BMP Effectiveness IV

A. Issue: Need to determine the cause and effect of management on the physical, chemical, and biological integrity of streams.

B. Issue: Determining the need to modify BMP's.

C. Issue: Need to develop data and information to respond to a law suit.

Skills Needed

The professional skills needed to monitor effectiveness increase with level. Likewise, the number of professionals and technicians needed increases as well as the number and types of expertise. An agency may have the skills needed at low levels, but may not have them at higher levels. Interdisciplinary monitoring is usually required and may require interagency cooperation to do the monitoring. Only professional, qualified, trained, or supervised people should be involved in effectiveness monitoring.

BMP Effectiveness I

Needed: One professional, one professional and a technician, or two professionals. They must be trained in the empirical/qualitative methods used, have the experience and knowledge to judge whether BMP's are clearly effective or not, and be able to discern when uncertainty occurs (called a grey area) and higher level methods and analyses are needed. Two people are often needed to take some measurements and record.

BMP Effectiveness II

Needed: Two professionals, trained in hydrology, fisheries, benthic invertebrates, and habitat, plus technicians. They must be trained and experienced in the qualitative methods used and the analysis procedures, have the experience to judge from the qualitative data whether BMP's are effective or not, and be able to discern grey-area situations. Interdisciplinary and interagency cooperation may be needed to bring together the necessary skills.

BMP Effectiveness III
Needed: Professionals in statistics, hydrology, fisheries, benthic invertebrates, aquatic habitat, channel geomorphology, and other professions. Which of these will be needed depends upon the situation and the stream/ecosystem/upland elements being monitored. Quantitative assessment clearly requires interdisciplinary effort and may require interagency cooperation to bring together the professionals needed.

BMP Effectiveness IV
This level of monitoring requires the same professionals as BMP Effectiveness III, but these people will likely be researchers from various agencies, universities, and other organizations.

Reference Condition

Reference streams or stream segments serve as controls and are essential to determining the effectiveness of management or BMP's in meeting the physical, chemical, and biological integrity of water. Reference streams are being used by many State water-quality agencies to define the reference condition for biological criteria for water-quality standards. Reference streams are necessary whether or not biological criteria have been established by the State. If the State has not established reference streams for the ecoregion in question, then the monitoring project plan and design must include the selection and establishment of one or more reference streams and a similar number of study streams. For the latter situation, the classification and selection of reference and study streams ought to be done with the full cooperation and support of the State water-quality agency.

The reference condition defines the range of biotic, habitat, chemical, and physical variables in reference streams. Biotic, habitat, and other water-quality variables vary temporally and spatially in reference streams. Several reference segments (10+) along a reference stream or in several separate reference streams are assessed for their physical, chemical, biological, and habitat condition. The data are analyzed to determine the variability of water-quality variables, thus the reference condition.

Reference condition should be determined by a professional biologist in cooperation with other specialists. It is not a simple task. In defining reference conditions, experience has found they are not just a product of abiotic conditions. For example, even with the best physical habitat, the site must be assessed and the biotic community data analyzed to determine if the site contributes to defining the reference condition.

The reference condition should be established using the highest monitoring level possible. The ability to define classes of water quality and ranges in parameters and to set water-quality criteria depends on methods used, number of samples, sample locations, accuracy of measurements, metrics, and analysis procedures. Using a high-level monitoring may yield more classes of water quality, a narrower range of reference conditions, and better water-quality criteria than using a lower level with simpler methods.

It is important to consider reference streams and their role in water-quality management under the Clean Water Act, EPA guidance to States, and in developing biological criteria. The expression "biological integrity" is used in the Clean Water Act to define the Nation's objective for water quality (Karr and Dudley, 1981) (EPA 1990). EPA (1990) defines biological integrity as follows: "According to Webster's New World Dictionary (1966), integrity is, 'the quality or state of being complete; unimpaired.' Biological integrity has been defined as 'the ability of an aquatic ecosystem to support and maintain a balanced, integrated, adaptive community of organisms having a species composition, diversity, and functional organization comparable to that of the natural habitats within a region' (Karr and Dudley 1981). For the purposes of biological criteria, these concepts are combined to develop a functional definition for evaluating biological integrity in water-quality programs. Thus, biological integrity is functionally defined as "the condition of the aquatic community inhabiting the unimpaired water bodies of specific habitat as measured by community structure and function." Ecological integrity is attainable when physical, chemical, and biological integrity occur simultaneously (Figure 4) (EPA 1990). Biological integrity includes habitat.

EPA (1990) recognizes the difficulty in finding unimpaired water to define biological integrity and to establish reference conditions. However, the structure and function of aquatic communities in high-quality streams can be approximated in several ways. First, characterize aquatic communities in the most protected waters in the ecoregion. In regions where few or no unimpaired sites are available, characterize the least impaired systems. In greatly disturbed regions, least impaired streams are those with the least-impact from human activities. Because least-impaired systems will be used as references, the limitations on biological, chemical, and physical integrity can be considered and incorporated into goals for those waters and the program to improve water quality.

EPA is using Omernik's (1987) ecoregion procedure for grouping streams. Within these ecoregions, reference streams are being selected and assessed to characterize the biological, physical, and chemical integrity for water quality criteria. Omernik's system uses an ecoregion framework for interpreting spatial patterns in State and national water quality data. His ecosystem classification is based on regional patterns in land-surface form, soils, potential vegetation, and land use. He recognizes the need to refine the classification within the broad ecoregions by considering stream size, hydrologic regime, and riparian vegetation. In addition to Omernik's (1987) factors for grouping streams in ecoregions, there are several more factors that can influence the physical, chemical, and biological integrity of water (see Table 4).

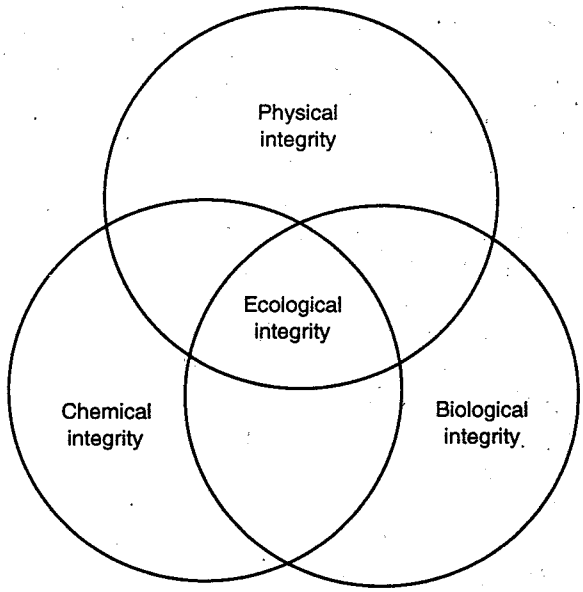


Figure 4. Ecological integrity is attainable when chemical, physical, and biological integrity occur simultaneously (EPA 1990).

Table 4. Factors that can influence the physical, chemical, and biological integrity of water

1. Prior land use
2. Stream order or size
3. Drainage area
4. Geology, soils
5. Stream gradient
6. Sinuosity
7. Aquatic vegetation and cover
8. Vegetation on slopes
9. Riparian vegetation
10. Amount of canopy over or shading stream
11. Sensitivity of watershed to change
12. Aspect
13. Fishing pressure
14. Hyporheic zones
15. Wetlands adjacent to streams
16. Elevation
17. Beaver dams and ponds
18. Water withdrawal or augmentation
19. Presence or absence of large woody debris
20. Volume and quality of ground water
21. Substrate type and heterogeneity
22. Substrate stability
23. Acid rain
24. Introduced aquatic species

Understanding how these factors influence the physical, chemical, and biological integrity of streams can help to fine tune classification of reference and study streams. For example, sediment from past abusive land use has been stored in stream channels and flood plains, but is now being removed from storage. Attached to these sediments are pollutants from past discharges. Also, prior land use affects the quality of ground water being discharged to streams today.

Fish and benthic invertebrate communities are influenced by stream size, stream order, watershed area, elevation, stream gradient, and substrate type, stability, and heterogeneity. Organ energy and functional feeding-group communities are influenced by the type of detritus from riparian vegetation, the amount of the stream's solar exposure, the presence of large woody debris, the type of detritus inputs from ephemeral streams from vegetated slopes, and the type and heterogeneity of substrate. The physical and chemical character of water is affected by geology, soils, riparian vegetation, slope vegetation, presence of wetlands, and water addition and withdrawal. Geology and soils influence the sensitivity of watersheds to change and affect landslide risk. Beaver dams and ponds change the hydrology and water quality characteristics of streams.

National and State efforts to classify streams by ecoregions need to be refined to focus on the influences affecting the physical, chemical, and biological integrity in the forest environment. A refined stream/ecosystem classification will eliminate a lot of the "noise" in the system and minimize the variation in conditions being compared between the reference and study streams. A good understanding of the forest stream/ecosystem will lead to better monitoring design, better data collection, better interpretation of data, and better water-quality management.

Barbour, M. T. et al. (1991) discuss the role of stream classification in bioassessment. "One function of classification is to increase the resolving power or sensitivity to biological surveys to detect impairment by partitioning variation within selected environmental parameters or among sites. The importance of minimizing variation...Clearly it is easier to distinguish impairment if the parameters have low variability. Formal statistical tests (parametric and nonparametric) indicate greater resolution and power exist if there is low variance within elements being compared. Effective classification leads to improving resolving power by partitioning or accounting for variability. A coarse classification yields higher variance and therefore lower resolving power; vice versa for finer classifications."

The USDA Forest Service is refining stream/ecosystem classifications for streams and watersheds on national forests. This effort will lead to finer stream classifications than Omernik's for ecosystem and water resource management on national forests. This refined classification may also be useful to State water-quality agencies in classifying some of their streams and ecoregions.

Stream/ecosystem classifications need to reviewed and refined after streams and watersheds have been subjected to major streamflow events that alter stream channels. To re-establish reference conditions after major storms, reference and study stream segments may need to be reclassified to ensure similarity and detect different responses between reference and study streams subjected to the same storm.

There are three ways that reference streams can be used in monitoring:

1. Reference Reach—A stream reach that has the same stream type and is in similar condition as the study reach being monitored for management impacts is called the "reference reach" and serves as the control. Reference and study reaches need to be in neighboring watersheds. It is assumed that the same storms, snowmelt rate, etc. and resulting streamflow will prevail on both the reference watershed and reach and the study watershed and reach (Figure 5).

2. Above vs. Below—Reference reaches on the same stream above a management activity are compared to study reaches below (Figure 6). Reference and study reaches need to be in near each other. Measurements are taken concurrently on both reaches.

3. Before vs. After—Calibrating of individual reach conditions before implementing a potentially impacting activity establishes a "baseline" against which the effect of the activity can be measured (Figure 7). Ideally, the period of evaluation will span similar weather patterns (dry vs. wet years, unique storms, etc.)

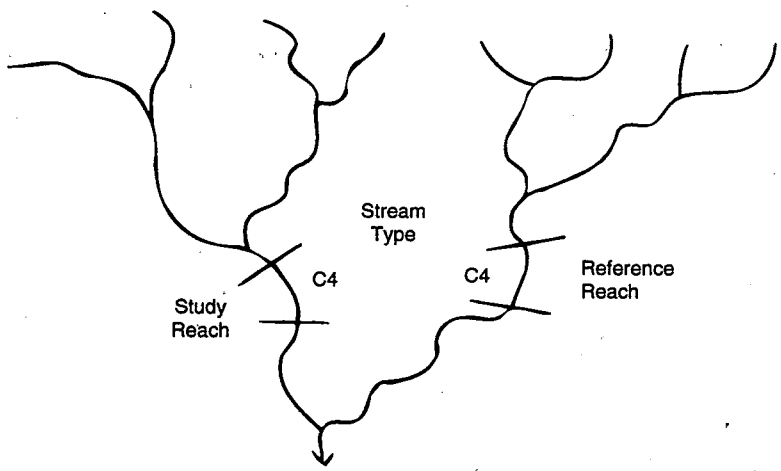


Figure 5. Reference reach channel classification similar to study reach.

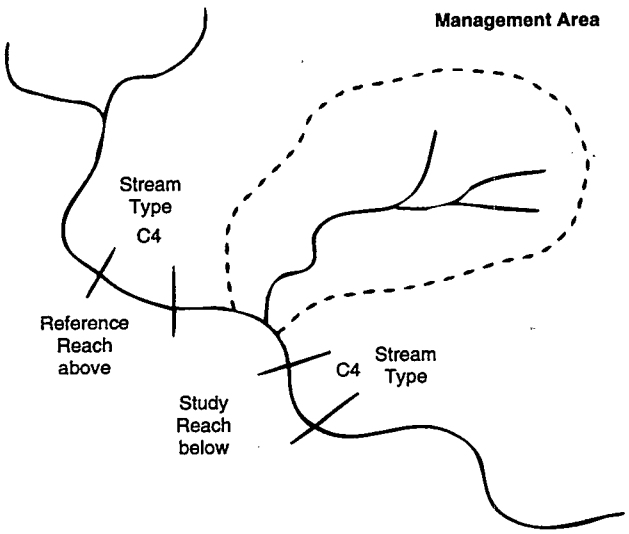


Figure 6. Above and below monitoring using reaches with the same stream channel classification.

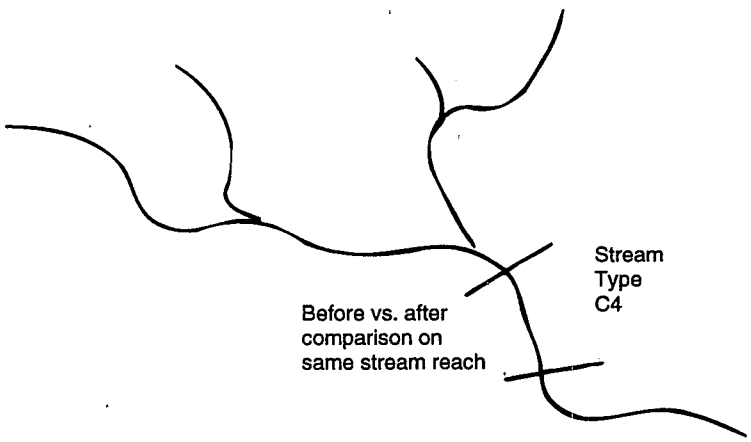


Figure 7. Before and after comparison on the same stream reach.

Reference streams are usually in separate watersheds from the study streams, especially for establishing biological criteria. However, reference stream segments above a management activity can be used to evaluate study segments adjacent to and below the activity. Again, the reference and study segments should be as similar as possible. Several reference segments representing least-disturbed conditions should be assessed to define a reference condition.

Guidelines for selecting reference and study streams vary by BMP effectiveness level:

BMP Effectiveness I

At this level of monitoring, there are several alternatives for selecting reference and study streams or segments:

1. If State reference streams for the ecoregion are similar to the proposed study streams, the empirical/qualitative monitoring can proceed directly to comparing the two.

2. If a State reference stream does not exist for the ecoregion, reference streams or segments similar to the study stream must be selected. A nearby stream in a least disturbed condition could be selected, assessed, and viewed to develop a reference condition for judging effectiveness. A mental picture of the habitat and biological health, and/or empirical evidence and qualitative data indicating the physical and chemical integrity of the reference stream would serve as the basis for comparison to the study stream.

3. A series of photographs of reference conditions can be compiled in a notebook and used for comparing study stream conditions: for example, pictures of stream types, riparian vegetation, stream bank stability, habitat types, and life forms associated with reference conditions. This tool would be useful in judging the effectiveness of BMP's and management practices for obvious yes-or-no in meeting water-quality integrity.

Reference segments selected above the managed area must represent least-disturbed conditions, not impaired waters. Making such a selection requires knowledge of watershed and stream conditions above the activity. To get this knowledge, the team would have to walk out the stream or use aerial photographs to judge whether the stream segment above is least impaired. Impaired segments must not be used as references, because (1) the impacts from above may obscure the impacts associated with the activity under study, (2) such references would define a low level of physical, chemical, and biological integrity as acceptable, which could lead to a management activity or BMP being judged as effective in meeting water-quality goals or standards when in fact it impairs the integrity of water.

The criteria to use in selecting references will depend upon the ecoregion and the key habitat needs of the aquatic species. Some general factors to consider include stream order, channel classification (for example, Rosgen's or Alaska channel classification, discussed under Channel Geomorphology

Monitoring Methods), type of riparian vegetation, etc. Using a few additional factors will lead to more similar streams being compared.

BMP Effectiveness II

The issue, monitoring design, and statistical requirements determine the number of reference and study streams, if the State has not defined reference conditions for the ecoregion. Qualitative and some quantitative data are collected for analysis of effectiveness. BMP Effectiveness II issues require detecting significant changes in the physical, chemical, and biological integrity of water. Therefore, the reference and study streams need to be similar to minimize variation and to increase ability to detect impacts of management. Reference and study streams must be in the same ecoregion. Additional classification factors to be considered include: stream size, channel classification (for example, Rosgen's or Alaska channel classification, discussed under Channel Geomorphology Monitoring Methods), type of riparian vegetation, type of vegetation of watershed slopes, type of substrate, and possibly aquatic habitat units.

BMP Effectiveness III

Again, the issue, monitoring design, and statistical requirements will determine the number of reference and study streams, which need to be as similar as possible. Stream similarity will be achieved using a detailed stream/ecosystem stratification system. If key habitat elements are known for intolerant aquatic life forms, these can be used in stratifying streams. If the key habitat elements need to be determined, the stratification will need to be detailed enough to minimize variation and allow identifying the key habitat elements from the data. Experts in the various aquatic life forms can help in selecting key factors to use in the stratification process.

A statistician needs to be involved in monitoring and can assist in determining the number of reference and study streams needed, which will depend on variability, level of change needed to be detected, and level of significance.

BMP Effectiveness IV

The stream/ecosystem stratification detail for this level is the same or slightly more detailed than for BMP Effectiveness III.

Amount of Data Collected

The amount and type of data needed vary significantly among monitoring levels, reflecting the number and complexity of methods used. It is important at all levels of monitoring that data be collected systematically according to a plan that specifies parameters. The observers must be properly trained to ensure consistent interpretation of conditions, measurements, and sample collection. Field, laboratory, and analysis forms need to be developed to ensure nothing is overlooked. Simple forms will suffice for low levels of monitoring, but will increase in complexity and detail with higher levels. Recording data on forms facilitates entry into a database. Data have lasting value only if stored in a database and can be readily accessed and analyzed. Analysis of a monitoring database can identify BMP's that are consistently effective, those that are only effective in

specific situations, and those that need modification and can reveal in-stream responses to BMP's implemented in various ecoregions.

Air quality assurance/quality control (QA/QC) program is needed at all levels of monitoring to ensure consistency of interpretation and accuracy of data. This includes checking performance of observers. It will vary by monitoring level and the variable being assessed. See Chapter 3 for more details.

Following is a general discussion of type and amount of data to be collected by monitoring level:

BMP Effectiveness I

Qualitative data will dominate, with some empirical observations. A small to moderate amount of data is recorded on a field form, which can be completed quickly. Recorded observations on reference and study streams will permit judging whether the BMP's are obviously effective or not in meeting water-quality goals or standards.

BMP Effectiveness II

Qualitative data will dominate, with a few measured observations. A moderate amount of data is collected.

BMP Effectiveness III

Quantitative data will dominate, with limited qualitative data. The amount of data collected can be large and the forms more complex.

BMP Effectiveness IV

Quantitative data will dominate, with limited qualitative data. The volume of data will be very large, reflecting the number and detail of monitoring methods used. The forms will be lengthy and complex.

Number of Streams Evaluated

The number of streams that can be evaluated will depend upon the time and resources available for monitoring. Given limited resources, as monitoring intensity increases from BMP Effectiveness I to IV, the number of streams and management activities that can be evaluated decreases significantly.

Length of Study

The study begins with planning monitoring and ends when a decision is made concerning BMP effectiveness in meeting water-quality goals or standards. Exact time frames cannot be predicted for each monitoring level, but general estimates can be made. The complete job includes defining the issues, selecting reference and study streams, designing and implementing the monitoring program, collecting and analyzing the data, entering data into a database, reporting results, and making the decision and recommendations.

BMP Effectiveness I

Usually a single site is visited to assess BMP effectiveness. Depending on the stream ecosystem classification used, the number of in-stream and on-slope observations made, and data and information analyzed, the time required may be a few hours to a week per site.

BMP Effectiveness II

Assuming more than one reference stream and study stream are being monitored, and depending on the amount of data collected per site, it may take 2 weeks to a month to reach a decision. In some cases, repeated visits to the sites may be needed to assess trends in recovery or to sample populations when they are at their peak. In such cases, the decision may be delayed.

BMP Effectiveness III

One-time visits may be made to sites on study streams for comparison with reference conditions. Even so, the amount of time to decision can be several weeks because of the number of streams, the detailed classification, the amount of data collected and analyzed, and detailed report preparation. Two to three months may be needed to do the complete job.

BMP Effectiveness III can involve monitoring management practices in several study watersheds and comparing them with several reference streams. One to three years may be required before decisions are made with this type of monitoring design.

A stream may be evaluated by segments or its whole length. The intensity of monitoring and length of streams will determine the time to decision.

When the stakes are high, decisions about specific management practices often need to be made quickly at this level. Detailed BMP Effectiveness III monitoring can be done by concentrating manpower and resources in a short period of time, thereby decisions can be reached in a few days or weeks.

BMP Effectiveness IV

Because this level usually involves research, the length of study will likely be at least 2 years.

Quality of Data for Decisions

The type and quality of data vary by monitoring level, methods used, and the skill of the people involved. Three types of data are taken: (1) empirical, (2) qualitative, and (3) quantitative. Quality is determined by the precision of measurements or observations.

The methods used at each level do not always clearly reveal whether goals or standards have been met. This uncertainty is termed the "grey area." As the monitoring level increases, the grey area decreases (Figure 8). Within these grey areas, the tendency is to assume the goals or standards have been met, even though higher levels of monitoring might reveal impairment. Uncertainty occurring with BMP Effectiveness I and II monitoring could trigger the use of higher levels of monitoring to resolve grey-area issues.

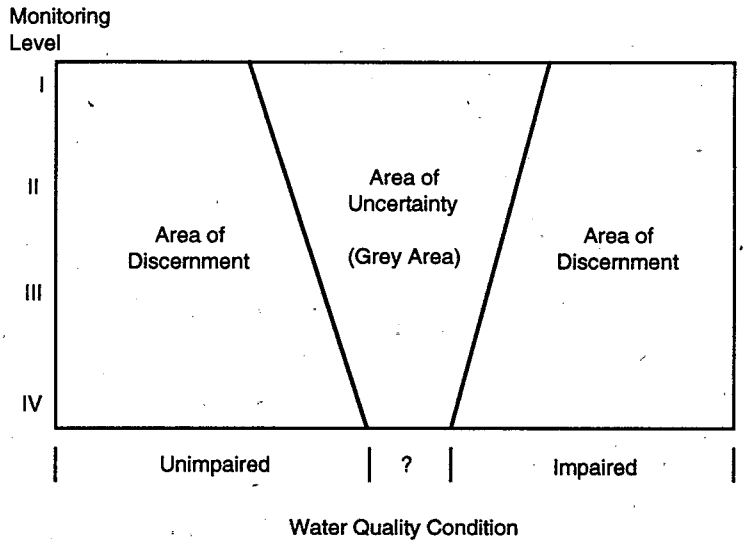


Figure 8. The area of discernment of water quality condition by BMP effectiveness monitoring level.

BMP Effectiveness I

Qualitative data dominates, with some empirical observations and quantitative data. Because BMP Effectiveness I relies on knowledge and experience gained from higher levels of monitoring, the observations and data should permit discerning when BMP's have clearly met water-quality goals or standards and clearly when they have not. However, the grey area may be large and should be recognized.

BMP Effectiveness II

Most of the data are qualitative, with moderate amounts of quantitative data. Level II monitoring should also benefit from knowledge and experience gained from higher levels of monitoring so the observations and data should be good for discerning when BMP's have clearly met water-quality goals or standards. The amount of data collected at this level is significantly more than for level 1. The precision of data also increases. The grey area is moderately sized. A wider range of effectiveness can be judged.

BMP Effectiveness III

Quantitative data with a high level of precision will dominate. Qualitative data are limited, but sound. The grey area is small, with most significant impacts detected.

BMP Effectiveness IV

Carefully collected quantitative data are coupled with a small amount of qualitative data. The precision is excellent. Methods used can detect major impacts and significant subtle changes in the physical, chemical, and biological integrity of water.

Quality of Decisions and Risk

There is risk in making the wrong decision about the effectiveness of BMP's. When the stakes are high, the risk needs to be minimized. Risk can be minimized by using a higher level of monitoring or recognizing the limits of the methods used at each level of monitoring. Usually the obviously effective and ineffective BMP's and management practices are easily recognized. The zone in between is the grey area and too often these situations are judged as effective, when they are not.

Decisions are subjective or objective. Subjective decisions, primarily based upon perception, judgment, and experience, should be backed by data and information derived from higher levels of monitoring. Objective decisions are based upon measurements, data, and analyses, although some judgment is involved.

BMP Effectiveness I

Decisions are subjective, with high risk in grey areas. The quality of information depends on the quality of training and skill of the observers. BMP Effectiveness I is most useful when BMP Effectiveness III and IV assessments are available to guide BMP Effectiveness I observations.

BMP Effectiveness II

Enough data are available to make objective decisions, yet judgment plays a major role in the decision. The grey area is reduced, thus reducing risk.

BMP Effectiveness III

Objective and informed decisions are based upon data. The risk is low and the decisions defensible.

BMP Effectiveness IV

Objective, data-based decisions are very defensible. The risk is low.

Monitoring Methods

Monitoring methods are organized into six categories: (1) on-slope, (2) chemical and physical, (3) channel geomorphology and stability, (4) biological, (5) habitat, and (6) aquatic vegetation (Table 5). The methods listed under each heading in Table 5 merely represent the techniques available. It is not the purpose here to present an exhaustive summary of methods. However, the list illustrates the types of methods needed to determine the effectiveness of BMP's in meeting water-quality goals or standards at various levels of effectiveness monitoring. The categories of monitoring methods in the table illustrate the types of questions that must be addressed to sort out water-quality responses to BMP's from other influences on water quality. Several items under each category may be needed to evaluate effectiveness.

Within each category, appropriate monitoring methods are identified for each monitoring level. The organization of methods by monitoring level is not rigid. For example, a monitoring plan may be oriented to a Level III assessment, but have one question addressed by a Level II method.

On-Slope Monitoring Methods

In-stream monitoring must be coupled with on-land monitoring to develop the linkage between what is observed in the stream and the management activity on the land. To evaluate NPS pollution from forest management requires evidence linking the activity to the stream impacts. So, on-land management activities need to be evaluated for their effectiveness in (1) preventing sediment, nutrients, pesticides, and organic debris from reaching the stream; (2) protecting stream banks; (3) shading the stream; (4) controlling surface runoff; and (5) minimizing soil exposure and compaction. In addition, the location of practices, the quality of installation, the timing of management activities, the appropriateness of BMP's selected for the site, the completeness of BMP's implementation on the site, and the timing of BMP implementation need to be considered.

The effectiveness of one set of management practices versus another can be estimated by using available models, installing plots and instrumented flumes on the slope, establishing sediment traps, and other methods. Such on-land comparisons reveal the relative movement of potential pollutants on the slope toward the stream. This information can identify management practices that will minimize the amount of sediment, nutrients, pesticides, and surface runoff from entering the stream.

On-Slope Methods by Monitoring Level

Comparing reference and study on-slope conditions is needed to link on-slope conditions with observations made in streams. Also, on-slope comparison between BMP's or management practices provides information on relative effectiveness of treatments.

For each effectiveness level, appropriate on-land monitoring methods are suggested. The page number beside each method identifies the location of the summary in the text.

Table 5. Monitoring methods by effectiveness monitoring level.

Monitoring Level.	On-Land Monitoring Methods	Chemical and Physical Methods	Channel Geomorphology Methods	Habitat Monitoring Methods	Biological Monitoring Methods	Aquatic Veg. Methods
BMP I	BMP Implementation I Erosion Indicators. Herbicide Indicators. USLE. Herbicide Cards. Florida Implementation Monitoring.	Empirical: clarity, color, sed. deposits, odor, scum, turbidity, and brightness/dullness of substrate. Field test kits. Grab samples. Current meters.	Empirical/perceptual. Pfankuch Channel Stability. Rosgen Channel Class. Pebble Counts	Empirical/ perceptual. RBP Habitat. Streamwalk. T-Walk.	Empirical/perceptual. Izaak Walton RBP I—Inverts. Streamwalk—Invertebrates. Hilsenhoff Biotic Index Vermont Guide—Inverts.	RBP I—Periphyton. RBP I—Aquatic Macrophytes. Montana Periphyton Protocol I
BMP II	BMP Implement. II. BMPEP. Erosion Indicators. USLE. Fabric Dams. Herbicide Indicators. Herbicide Cards.	Field test kits. Grab samples. Current meters.	Pfankuch Channel Stability. Rosgen Channel Class. V* Pool Sediment Pebble Counts	RBP Habitat. Basinwide Estimation of Habitat. Hankin - Reeves.	RBP II—Inverts. RBP V—Fish. Streamwalk—Inverts. Snorkel—Fish. Vermont Guide—Inverts. NC 4 Sample Streamwalk II	RBP II—Periphyton. RBP II—Aquatic Macrophytes. Montana Periphyton Protocol I or II
BMP III	BMP Implementation III. BMPEP. Fabric Dams. Flumes & Samplers. Herbicide Cards. Montana Implementation.	Temporal/spatial integrated samples. Analytical labs with EPA QA/QC. Ground Water. Raingages. Recording stream gages.	Pfankuch Channel Stability Rosgen Channel Class. plus channel geomorph. relationships. V* Pool Sediment Pebble Counts Rosgen Channel Monitoring	RBP Habitat. Hankin-Reeves. Bisson. R5-FHR. Platts Riparian Habitat. BASS Ohio-QHEI	RBP III—Invertes. RBP V—Fish. Hankin-Reeves—Fish. BASS—Inverts. & Fish. Ohio EPA—Inverts. & Fish. Snorkel—Fish. NC 4 Sample NC 10 Sample	RBP III—Periphyton. RBP III—Aquatic Macrophytes. Montana Periphyton Protocol II
BMP IV	BMP Implementation III. Fabric Dams. Flumes & Samplers. Rainfall Simulators. Herbicide Cards.	Temporal/spatial integrated samples. Analytical labs with EPA QA/QC. Ground water. Rain gages with chemical analysis. Recording stream Gages & Weirs.	Pfankuch Channel Stability. Rosgen Channel Class., plus channel geomorph relationships. V* Pool Sediment Pebble Counts Rosgen Channel Monitoring	RBP Habitat. Hankin - Reeves. Bisson. R5-FHR. Platts Riparian Habitat.	RBP III—Inverts. RBP V—Fish. Ohio EPA—Inverts & Fish. Hankin-Reeves—Fish. BASS—Inverts & Fish. NC 10 Sample	RBP III—Periphyton. RBP III—Aquatic Macrophytes.

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BMP Implementation I

At the BMP Effectiveness I level, one individual can inspect the management area to determine if the BMP's were properly selected and implemented. A short inspection form can be constructed reflecting the State BMP guidance or requirements. Using such a form will ensure that the inspection considers all appropriate BMP's for the management situation. Inspection observations are recorded on the form for later reference and for entering into a database. The database can be analyzed to determine trends in BMP compliance and identify those BMP's that have poor compliance.

The observer should note any specific BMP's that were not properly implemented and those that appear to have failed. If fresh sediment is found in the stream, the observer should attempt to trace it to its source to determine if it came from the management area.

Florida's BMP Implementation Monitoring

A good example of BMP Implementation I monitoring is Vowell (1990). He reports that in 1981 the Florida Division of Forestry (DOF) developed the Best Management Practices compliance survey to help direct BMP implementation in the State. The survey, which is conducted every 2 years, was designed to (1) measure the compliance of forestry operations with State-approved BMP's, and (2) identify specific implementation and practice deficiencies. Vowell describes the Florida BMP Compliance Survey, including (1) the method of site selection, (2) the field procedures, and (3) follow-up activities:

Site Selection

During a survey year approximately 150 individual forestry operations (survey sites) are selected by county for compliance monitoring. The number of sites per county is based on the USDA Forest Service Timber Survey estimated of harvest levels. Survey sites are identified from aerial observations, using fixed wing aircraft. Aerial site identification helps maintain a more representative cross section of ownerships, forest types and physiographic regions by minimizing bias due to ground accessibility, and by providing a more random site distribution within each county.

Survey sites are selected on the basis of two criteria: (1) length of time since the forestry operation took place, and (2) proximity of the operation to a body of water. In order for a site to qualify for the survey, the forestry operation must have occurred within 2 years and be located within 300 feet of an intermittent or perennial stream or a lake 10 acres in size or larger. The purpose of these criteria is to limit the number of sites to those in which the greatest potential for impacts exists, and to increase the probability that any impacts resulting from forestry activities are still visible.

Data Collection

Once a survey site has been selected, a trained professional forester visits the site and evaluates the operation for compliance with BMP's. The evaluation involves a detailed field inspection and completion of an 85 point questionnaire. The questionnaire, which focuses on roads, streamside management zones and site preparation, was designed with several objectives in mind. First, to encourage the observer in the field to "get out and look" rather than conduct a "windshield survey." In responding to the questionnaire, the observer will consider virtually every aspect of the operation that took place as well as the local topography, soils, timing, ownership and whether or not there was technical assistance involved. However, as with any exercise of this nature there are always some "gray" areas which can result in subjective judgment. Hence, the questionnaire consists primarily of yes/no and multiple choice type questions, which minimize the potential for biased re-sponses and also makes the data more compatible for computer analysis. The final entry in the questionnaire asks for an overall "pass/fail" rating of the site. This rating is based solely on the information from the questionnaire and takes into consideration all aspects of the operation rather than any single and/or perhaps more visible practice or problem.

Division of Forestry
Department of Agriculture & Consumer Services
BMP Compliance Check

Survey Number_____

I. General Information

1. County_____ Section_____ Township_____ Range_____
2. Approx. No. of acres treated (logged, planted and or site prepared)_____
3. Ownership (check one) ____Government, ____Industry, _____Private non-ind.
4. If private non-industrial, was technical (professional) forestry assistance provided by (check one): ____DOF Forester, ____Consultant, ____Industry Forester, ____None at all.
5. Dominant site type (before treatment): ___Natural pine, ___Pine Plantation, ____Mixed Pine/Hardwood, ____Bottomland Hardwood, _____Upland Hardwood, ____Field or Pasture.
6. Type of Treatment (check one or more): ___Clearcut, ____Selective cut or shelterwood, ____Seed tree cut, ____Site preparation for natural regeneration,____Site preparation for hand planting, ____Site preparation for machine planting,____Chemical site preparation.

II. Various Site Characteristics

1. Terrain: ____Bottomland, ____Flatwoods, ____Upland Clay Hills, ___Sandhill.
2. Principle soil texture: ____Clay, ____Loam, ____Sand.
3. Highest slopes approaching stream bottoms: ____%.
4. Soil type (from soil survey, if available)_____.
5. Soil erodibility (K factor): ____High, _____Medium, _____Low.

III. Water Conditions in Area

1. What type of water body (bodies) is on or border the site: ____Perennial stream, ____Intermittent stream, _____Lake/pond (10 acres or greater in size), ____Canal, ____None of the above. Explain: (Name of water body if known)_____
2. How close did the following activities get to the water's edge, if applicable? Site preparation____feet, Clearcut_____feet, Mechanical tree planting____feet.
3. Evidence of damage to stream channel: (may check more than one) ____ Debris left in stream channel from temporary crossing.

_____ Slash, other debris randomly left in stream.
_____ Erosion and/or failure of stream channel banks.
_____ Sediment is being deposited in stream.
_____ None
_____ Other (specify)_____

_____

IV. Roads/Skid Trails

	YES	NO
1. Roads systems on site. If no, then go to #7/	_____	_____
2. Does any newly constructed road system (as a result of the operation) occur within 300 ft. of a watercourse?	_____	_____
3. Is there sediment from any road system being deposited into the watercourse?	_____	_____
4. Are road systems generally located to avoid steep slopes and gullies?	_____	_____
5. Are roads locate to avoid streams, and depressional areas as much as possible?	_____	_____
6. Are unneeded access roads closed to vehicular use?	_____	_____
7. Are any stream crossings on site? If no proceed to #12.	_____	_____
8. Are stream crossings adequately stabilized?	_____	_____
9. Is harvesting equipment (skidders) randomly crossing streams?	_____	_____
10. Is there sediment being deposited into stream from skid trails?	_____	_____
11 Are planned stream crossing made at right angles to the watercourse?	_____	_____
12. Any culvert or bridge washouts.	_____	_____
13. Are any roadside ditches pulled (connected) directly into watercourse?	_____	_____
14. Were oil and trash properly disposed of?	_____	_____

15. Evidence of BMP's applied to roads (check one or more)
    _____ Waterbars installed
    _____ Broad-based or rolling dips installed
    _____ Mulching, seeding, and/or fertilizing
    _____ Water turnouts or wing ditches
    _____ Effort made to minimize slopes on road
    _____ Roads exist, but no BMP's were applied to them
    _____ No roads on site
    _____ Other (specify)_____

    _____

16. Evidence of BMP's applied to skid trails:
    _____ Seeded

_____ Waterbars
_____ Slopes minimized
_____ Skid trails exist, but no BMP's were applied
_____ No skid trails apparent on site
_____ Other (specify) _____
_____

17. If new roads were constructed, who was responsible for layout and construction? (maybe more than one)
_____ Government crew
_____ Landowner
_____ Forest industry crew
_____ Private vendor
_____ Information not available
_____ Not applicable
_____ Other (specify) _____
_____

18. Were erosion control efforts on road systems generally effective and adequate? _____ Yes _____ No

Evidence is: (check one or more)
_____ Water is diverted off roads at intervals which are successfully controlling surface erosion.
_____ Water is diverted away from road prior to reaching stream channels.
_____ None installed, erosion occurring
_____ None installed, no evidence of erosion

19. Were landings and decks properly located, adequately cleaned up, and stabilized? _____ Yes _____ No

V. Site Preparation and Regeneration

1. Method of site preparation (check one or more): _____ Bulldozed, _____ Disk/harrow, _____ Shear, _____ Chop, _____ Burn, _____ Bed, _____ Root Rake, _____ Herbicide, _____ Not applicable, Other (specify)_____

	Yes	No	N/A
2. Was site planted or to be planted by hand?	___	___	___
3. Was site planted or to be planted by machine?	___	___	___
4. Planting follows natural land contours? If not, go to #8	___	___	___
5. Was slash and debris windrowed/piled?	___	___	___
6. Is there excessive topsoil pushed into windrows/piles?	___	___	___
7. Do windrows follow natural land contours?	___	___	___
8. Were any new lateral drainage (field) ditches constructed?	___	___	___

9. Do established lateral field ditches occur on site? ___ ___ ___
10. Do firelines exist on the site? If no, go to #14. ___ ___ ___
11. Does erosion appear to be associated with firelines? ___ ___ ___
12. Any waterbars, other water diversions used on firelines? ___ ___ ___
13. Do firelines avoid streamside management zones (SMZ)? ___ ___ ___
14. If a perennial stream or lake (10+ acres) is adjacent was a selective cut 35' wide Primary SMZ maintained throughout? (Indicate N/A if no perennial water is present) ___ ___ ___
15. If necessary (according to guidelines for perennial streams or lakes 10+ acres), does a secondary SMZ appear and does it meet guidelines? (Indicate N/A if no perennial water is present) ___ ___ ___
16. If an intermittent stream, was mechanical site preparation avoided within the required SMZ? (Indicate N/A is no intermittent water is present or if site had not site prepared). ___ ___ ___
17. Type of operator responsible for logging/site preparation work on this site:
    _____ Private landowner, using own equipment, _____ Government crew,
    _____ Vendor contracted by forest industry, _____ Forest industry crew,
    _____ Vendor contracted by private landowner, _____ Other (specify)
    _____
18. Additional comments/summary concerning this site:_____
    _____
19. General grade - Do you feel there was generally good compliance with 208 guidelines on this site? _____ Yes, _____ No. If not, briefly state primary reason for site failure._____
    _____
    _____

_____	_____
Forester	Landowner
_____	_____
Telephone	Address
_____	_____
Date	City, State, ZIP Code

BMP Implementation II

At the BMP Effectiveness II level, one observer visits the management area and evaluates in detail the BMP implementation and effectiveness. Roads and skid trails are carefully examined for waterbars, proper drainage, proper bridge and culvert installation, cut-and-fill slope revegetation success, proper location, etc. For example, the effectiveness of waterbars on a skid trail would be evaluated by tracing the path of runoff below the waterbar to determine if the runoff reached the stream and discharged sediment into it. As another example, runoff and eroded soil from a site-prepared area could be traced through the filter strip to determine if runoff and sediment reached the stream. If fresh sediment is found in the stream, the observer would attempt to trace it to its source and determine if it was caused by a specific practice or disturbance in the management area.

BMP Implementation III

This method is applicable to both BMP Effectiveness III and IV assessments. An interdisciplinary team conducts a detailed inspection of BMP implementation and effectiveness as outlined in BMP Implementation II. An interdisciplinary team may have a forester, a road engineer, a logging engineer, a State water-quality representative, a soil scientist, a hydrologist, and/or a fisheries biologist. The team can be interagency, too. Each member evaluates specific BMP's in terms of his or her expertise, and records his/her compliance and effectiveness observations on a field form. Team members assemble at the end of the inspection, discuss their observations, and come to a consensus as to the level of compliance, the quality of implementation, and the effectiveness of BMP's. The consensus results are recorded on a form for entering into a database. The results of the team inspection should be discussed with the landowner and operators. The landowner and operator can provide information as to conditions at the time of the management activity, discuss the management plan and the problems they encountered, and explain why they chose to deviate from the plan. These discussions will help the team members interpret their observations and rate compliance and effectiveness.

Montana's Forestry BMP Implementation Monitoring

A good example of Level III implementation monitoring is Montana's Forestry Best Management Practices Implementation Monitoring (Schultz 1992). This process is used to:

1. determine if BMP's are being applied on timber harvest operations.
2. evaluate the general effectiveness of BMP's in protecting soil and water resources.
3. provide information on the need to revise, clarify, or strengthen BMP's.
4. provide information to focus future study efforts by identifying subjects and geographic areas in need of further investigation.

For the 1992 field audit, Montana used three audit teams, each assigned a region of the State (northwest, west central, and southwest/east regions of State) (Schultz 1992). Each team was composed of six members: a fisheries biologist, a forester, a hydrologist, a representative of a conservation group, a road engineer, and a soil scientist. One member of each audit team was assigned to be leader and was responsible for filling out the consensus rating form and for logistics. Team members were Federal, State, and industry employees, and consultants.

Implementation monitoring was limited to those timber harvests that were most likely to impact water quality. The initial criteria for site selection were:

1. Sites harvested in 1990, 1991, or 1992 (last 3 years). Priority was given to the most recently harvested sites.
2. Part of the harvest area needed to be within 200 feet of a stream.
3. Minimum size of harvest unit to audit was 5 acres. Minimum timber removal was 7,000 board feet per acre. Some flexibility was allowed if smaller areas or low-volume harvests met other criteria.
4. In aggregate, sites needed to reflect the distribution of timber harvest in Montana, geographically and by ownership. Sites were classified into four land-ownership categories: nonindustrial private, industrial, Federal, and State. The number of monitoring sites per type of ownership was proportional to the volume of timber harvested by each class in 1990, the latest complete recording period. At least five sites per ownership class were audited.

Sites that met the initial criteria were further stratified by the Montana Department of State Lands according to the following priorities (Schultz 1992):

1. Sites with road construction or reconstruction and slash disposal completed.
2. Sites with road construction or reconstruction but slash disposal not completed.
3. Sites with slash disposal completed but no road construction.

Sites were further stratified by erosion-hazard. Two-thirds of the audits were located on high-erosion-hazard sites; one-third were located on low-to-medium-erosion-hazard sites. This stratification was subject to availability of sites in the various erosion-hazard classes. Erosion hazard is a function of slope steepness and erosivity, based on geologic parent material and soil texture. Care was taken that a disproportionate number of high-hazard sites did not fall in any one ownership class.

The audit team's rating form covered 58 potential practices at each site, rating application and effectiveness of each practice on a 5-point scale (Schultz 1992). The team rated a BMP by noting first if it was applicable to the site, and if so, whether it was applied correctly and in the proper locations. Lack of adequate application or misapplication were regarded as departures from BMP's. The rating guide for application of BMP's was:

5—Operation exceeds BMP requirements.
4—Operation meets BMP requirements.
3—Minor departure from intent of BMP.
2—Major departure from intent of BMP.
1—Gross neglect of BMP.

Ratings of 5 and 4 are self-explanatory. The 3 rating applies to small departures distributed over a localized area or over a large area where potential for impact is low. The 2 rating applies to large departures or BMP's repeatedly neglected. The 1 rating applies where risks to soil and water resources were obvious and yet there was no evidence indicating that operators had applied BMP's to protect these resources (Ehinger and Potts 1990).

The effectiveness rating answered questions concerning impacts (Schultz 1992). For example, "Has the application or misapplication of a particular forest practice increased the likelihood or actual occurrence of surface sediment in the stream channel?" The rating guide for effectiveness was:

5—Improved protection of soil and water resources over pre-project conditions.
4—Adequate protection of soil and water resources.
3—Minor and temporary impacts on soil and water resources.
2—Major and temporary, or minor prolonged, impacts on soil and water resources.
1—Major and prolonged impacts on soil and water resources.

Before the audit, the work group defined the above terms prior to facilitate consistency:

Adequate—Small amount of material eroded; material does not reach draws, channels, or flood plain.
Minor—Some material erodes and is delivered to draws but not to stream.
Major—Material erodes and is delivered to stream or annual floodplain.
Temporary—Impacts lasting 1 year or less, no more that one runoff season.
Prolonged—Impacts lasting more than 1 year.

The teams conducted the audits July through September of 1992. Usually a representative of the landowner briefed the team by giving background information on the silvicultural prescription, time of operation, and associated practices. Team members walked the site, reviewing potential impact areas such as roads, streams, streamside management zones, skid trails, and firelines. Teams typically spent about two hours inspecting each site.

After finishing the inspection, the team gathered to discuss the rating while still on site. Each team completed a single rating form for each site. If the team could not reach consensus, the rating was determined by vote. The team leader noted dissenting opinions in the comments section. The results were discussed with the landowner representative, if available.

The results of the 1992 audits are summarized below (Schultz 1992).

Application of BMP's (All Ownerships Combined)

Meets or exceeds	87%
Minor departures	8%
Major departures	3%
Gross neglect	2%

Effectiveness of BMP (All Ownerships Combined)

Adequate protection	90%
Minor/temporary impacts	6%
Major/temp, Minor/prolong	3%
Major/Prolong	1%

Best Management Practices Evaluation Program—BMPEP

Roby, Rector, and Furniss (1992) have developed an interdisciplinary, detailed BMP implementation/effectiveness monitoring methodology for the California Region, USDA Forest Service that is applicable to both BMP Effectiveness II and III levels. Through a iterative process of BMP implementation, monitoring BMP compliance and effectiveness, and BMP development/modification, BMPEP strives to improve BMP's so that water-quality goals will be met.

The objectives of the BMPEP are to:

1. Assess the degree of implementation of BMP's.
2. Determine which BMP's are effective.
3. Determine which BMP's need improvement or development.
4. Fulfill Forest Land and Resource Management Plan BMP monitoring commitments.
5. Record performance of NPS pollution management in the California Region.

The BMPEP has three primary components:

1. Administrative Evaluations
2. On-Site Evaluations
3. In-Channel Evaluations

Administrative Evaluations are broad-scale, subjective assessments of multiple BMP's at the project level. These assessments look at the total project, from the planning and design phases to implementation, and subjectively evaluate effectiveness of the resulting project in attaining desired water-quality protection. The assessment evaluates factors not observable or measurable at a project site, such as the Interdisciplinary Team (ID) process and communication during project development and execution.

The Interdisciplinary Team process is normally evaluated by members of review teams during project and activity program reviews. The review assesses the process used on the Forest to identify and schedule harvest activities, evaluates the quality of ID teamwork and NEPA documents, and rates how well the process is being used and how well it is working.

On-site Evaluations provide a means of gathering objective data at the site of BMP implementation for specific practices. The evaluations are based on actual measurements of key criteria (ground cover, canopy closure, etc.) and ocular estimates (presence or absence of rills, debris in culvert inlets, etc.)

There are 28 different On-Site Evaluation Procedures, each dealing with one of the following:

1. Streamside-management zones (SMZ)
2. Skid trails
3. Suspended yarding
4. Landings
5. Timber administration
6. Special erosion control and revegetation

7. Meadow protection
8. Road surface, drainage, and slope protection
9. Stream crossings
10. Control of sidecast material
11. Servicing and refueling
12. In-channel construction practices
13. Temporary roads
14. Rip rap composition
15. Water-source development
16. Snow removal
17. Pioneer road construction
18. Restoration of borrow pits and quarries
19. Protection of roads during wet periods
20. Designated swimming areas
21. Developed recreation sites
22. Location of stock facilities in wilderness
23. Range management
24. Prescribed fire
25. Mining operations
26. Common variety minerals
27. Vegetation manipulation
28. Revegetation of surface-disturbed areas

A detailed assessment of BMP implementation is conducted. Rating involves reviewing project plans, environmental assessments, and the actual practices on-the-ground to gauge how well the implemented practices match what was planned.

The extent of BMP effectiveness is evaluated based upon observation on the slope. Based on experience and judgment, the effectiveness is rated subjectively by such comments as:

- Effects are to a perennial stream and are major and prolonged ("major" meaning that beneficial uses of water could be affected; and "prolonged" meaning more than a year).

- Effects are to an intermittent or ephemeral stream and are major and prolonged.

- Effects are to a perennial stream and are either minor and prolonged or major and temporary.

- Effects are to an intermittent or ephemeral stream and are either minor and prolonged or major and temporary.

- Effects are to perennial stream and are minor and temporary.

- Effects are to an intermittent or ephemeral stream and are minor and temporary.

- Effects extend to the SMZ, but not to the channel of a perennial stream.

- Effects extend to the SMZ, but not to the channel of an intermittent or ephemeral stream.

The data are recorded on a field form and entered into a database for subsequent analysis and report writing. The skid-trail form is an example of the forms developed for this method (Figure 9).

Erosion Indicators

Evidence of sheet erosion, the amount of bare soil, the presence of rills, gullies, and slumps, rutting, and compaction are on-site indicators of management and BMP effectiveness. Several factors affect surface-erosion rates in addition to amount of bare soil: slope, slope length, rainfall energy, and the inherent erodibility of the soil. When comparing the effect of two different practices on erosion, the two areas should be similar in slope, soils, rainfall, and slope length. The amount of bare soil can be significantly different. However, a large amount of bare soil on a gentle slope may yield as much erosion as a small amount on a steep slope.

On-site erosion, as evidenced by erosion pedestals, is not usually visible until erosion rate exceeds 10 tons per acre per year or approximately 1/16th of an inch of soil. Erosion pedestals capped with small stones, leaves or twigs measuring a 1/16th inch and shorter are difficult to detect. This explains why many observers see no evidence of erosion when in fact significant sheet erosion is occurring.

On-site detachment, as evidenced by erosion pedestals, does not automatically mean that all detached soil leaves the slope; most detached soil is trapped/stored on the slope before it reaches the toe of the slope.

However, if erosion pedestals are clearly evident, the effect of two treatments on erosion can be compared, using the rule-of-thumb that an area inch equals approximately 160 tons per acre. If one management area has 50 percent bare soil and the average pedestal height is 1/4th inch in the bare areas, on-site soil detachment is estimated at 20 tons per acre (160 tons/acre inch x .25 inch x 0.5). If a second area of the same age received about the same rainfall, has 15 percent bare soil, and pedestal height averages 0.2 inches, the on-site detachment for the second area is approximately 5 tons per acre.

The presence of rills or gullies suggests significant surface runoff and movement of eroded soil down the slope. Rills are rivulets on the slope generally less than one square foot in cross section; rill paths generally are parallel. When rills converge or the cross sectional area exceeds one square foot, they become gullies. The frequency and size of rills and gullies on the slope can be used to compare erosion between management areas.

The amount of rill and gully erosion area can be approximated by estimating the percent of the area occupied by rills and gullies and their average cross sectional area. For example, if rills occupy 5 percent of the area and have an average cross sectional area of 16 square inches, the rill erosion rate is estimated at about 10 tons per acre: ((.05 X 43560 ft^2/acre X 85 lbs/ft^3 of soil X 16 in^2/144 in^2)/(2,000 lbs/ton) = 10.3 tons/acre). The same method can be used for estimating small landslides and slumps. The total estimated erosion on a site would be the sum of sheet and rill erosion, plus measured gully erosion, plus measured landslide erosion. If the time since management began is known, rill and gully erosion rates per year can be estimated.

Best Management Practices Evaluation

UTM Coordinates Zone __ __
Easting __ __ __ __ __ __
Northing __ __ __ __ __ __

Form T02: Skid Trails
(BMP 1.10 & 1.17)

ID#: _____
Selection Code: _____

Reviewer(s) _____ Title(s) _____ Date _____ Forest _____ District _____
Project _____ Unit # _____ Year Logging Occurred _____ T _____ R _____ S _____
Rock Type _____ NFS Watershed _____

IMPLEMENTATION

Skid Trails comply with FSH standards, and any special EA conditions as they relate to: **Rating**

a) Location ..... ____
b) Drainage and Erosion Control ..... ____
c) Width ..... ____
d) Drainage Crossings ..... ____
f) Endlining *(if req. by TSC C.6.422)* ..... ____

1 = Exceeds contract/project requirements
2 = Meets contract/project requirements
3 = Minor departure from contract/project requirements
4 = Major departure from contract/project requirements
Rate as NA if criteria not applicable at this site

If any Implementation Rating is "3" or "4", complete the following:
Problem occurred in which phase(s) of the project: ☐ Site Evaluation ☐ Plan Prescription ☐ EA/EIS ☐ Contract ☐ Layout ☐ Administration
Describe deficiencies and corrective actions:

EFFECTIVENESS

1) Ground Disturbance	☐ Skid trails disturb less than 10% of unit	☐ Skid trails disturb > 10% but < 15% of unit	☐ Skid trails disturb > 15% of unit
2) Erosion on skid trail surface	☐ Little or no evidence of rills	☐ Rills present, but occur on < 20% of skid trail surfaces	☐ > 20% of surface has rills, or rills present that are > 2" deep and > 10' long
3) Rutting	☐ Little or no evidence of rutting	☐ Some rutting present, but < 10% of area has ruts > 2" deep	☐ > 10% of surface length has ruts >2" deep
4) Waterbars			
a) Diversion of runoff	☐ < 10% of waterbars fail to divert flow off of skid trail	☐ >10% but < 20% of waterbars fail to divert flow off skid trails	☐ > 20% of waterbars fail to divert flow from skid trail
b) Sediment below outlet	☐ Sediment deposition absent or does not extend beyond outlet control	☐ Sediment deposition evident but does not extend > 20' below waterbar outlet	☐ Sediment deposition extends > 20' below waterbar outlet
c) Erosion below outlet	☐ No evidence of rills or gullies	☐ Rills present, but < 20' long or occur on < 20% of waterbar outlets	☐ Rills > 20' long or occur on > 20% of waterbar outlets
d) Sediment to channel	☐ No evidence of transport to SMZ	☐ Sediment deposited in SMZ but not in channel	☐ Evidence of sediment transport to or deposition in channel

over

Form #T02 Version 4/92

Form T02: Skid Trails *(Page 2)*
(BMP 1.10 & 1.17)

EFFECTIVENESS *(continued)*

FOR SITES WITH STREAM CROSSINGS:

5) Sediment to Channel: stream crossing rilling	☐ Rills may be evident, but are infrequent appear stable, with no evident sediment delivery to channel	☐ Rills present, but average less than 1 per 5' lineal, rills not enlarging. Minimal evidence of deposition in channel. No gullies	☐ Numerous rills present (>1 per 5' lineal) apparently active or enlarging, evidence of delivery to channel, or gullies present

If poor effectiveness is evident, comment on:
(1) Possible causes *(e.g., site sensitivity, inadequate BMP prescription, major storm event, etc.)*:

(2) The degree and duration of effects on beneficial uses of water:

Figure 9. Example form developed for the BMPEP monitoring program.

Universal Soil Loss Equation

For sheet and rill erosion, the Universal Soil Loss Equation (USLE) has been adapted to forest situations in Eastern and Western United States, and is applicable for BMP Effectiveness I and II levels. The USLE estimates soil erosion from rainfall but not snow melt.

Despite being called "Universal," the equation cannot be applied to new areas without adaptation. Vegetation characteristics, soils, rainfall, and other conditions vary among regions of the country. The factors and subfactors in the USLE can be modified to increase the accuracy of erosion estimates for local conditions (Dissmeyer 1982a and 1982c, and Dissmeyer and Foster 1981). This has been done for the Eastern United States.

The USLE predicts erosion on watershed slopes from the top of a slope to level ground where deposition occurs, or where the surface runoff flows into a depression, road ditch, skid trail, gully, or stream. In the forest environment, deposition often occurs at the junction of a management area and the filter strip, which defines the end of the slope for the USLE and where erosion stops.

Erosion estimates made with the USLE do not be reflect the amount of sediment reaching a stream or the sediment yield from a watershed. In most forest situations only a small portion of the sheet and rill erosion reaches a stream. Much of the eroded soil is trapped behind debris, in litter on the slope, or in intervening vegetation. It is important to trace soil movement down slope to determine it runoff has discharged sediment into a stream. Only a portion of the management area will yield sediment. Using the area contributing sediment and observations on the amount of deposition on the slope, skilled observers can estimate the percent of erosion reaching a stream. This is very subjective, however, and requires a great deal of experience and testing estimates against measured sediment yields.

The Wischmeier and Smith (1978) USLE procedure significantly overestimated erosion for undisturbed and managed forest situations in the Eastern United States. This led to a modification by Dissmeyer and Foster (1984). Their methodology was tested on 39 research watersheds treated with a wide range of common forestry practices, from undisturbed to logging to the most intensive mechanical site preparation for tree planting. The results confirm that their procedure gives reasonable estimates of erosion on managed forests. The Dissmeyer and Foster (1984) modification is as follows:

The Universal Soil Loss Equation is: A = RKLSCP
Where:

***A** is the computed soil loss per unit area, expressed in units selected for **K** and for the period selected for **R**. In practice, these are usually so selected that they compute **A** in tons per acre per year, but other units can be selected.*

***R**, the rainfall and runoff factor, is the number of rainfall erosion index units. The **R** factor is usually read from an average annual rainfall index map. For specific years or locations, the **R** factor can be computed from recording rain gage records.*

__K__, the soil erodibility factor, is the soil loss per unit of __R__ for a specific soil as measured on a unit plot, which is defined as a 72.6 foot length of uniform 9 percent slope continuously in a clean-tilled fallow. Clean-tilled fallow is an agricultural condition where the soil is kept bare and disked up and down slope—the most erosive condition for agricultural land. The __C__ cover management factor has subfactors that adjust __K__ to reflect untilled forest management conditions.

__L__, the slope-length factor, is the ratio of soil loss from the field slope length to that from a 72.6 foot length under identical conditions.

__S__, the slope-steepness factor, it the ratio of soil from the field slope gradient to that from a 9 percent slope under otherwise identical conditions.

__C__, the cover-management factor, is the ratio of soil loss from an area with specific cover and management to that from an identical area in tilled, continuous fallow.

__P__, the support practice factor, is the ratio of soil loss with a support practice like contour disking to that with straight-row farming up and down slope.

It should be noted that the USLE continues to be revised and improved, recently with improvements in the R, K, and LS factors (Reinard et al., 1991). A revised R-factor map better reflects both western and eastern conditions. The K factor now reflects seasonal differences in erodibility. The LS guidance has been revised to help the user better select slope lengths appropriate to field conditions. These revisions have refined erosion estimates with the Dissmeyer and Foster (1984) procedure.

The C and P are combined for forestry application and called the cover-management factor (CP) (Dissmeyer and Foster 1984). The forestry modifications to the USLE were modifications in the subfactors of CP and the addition of a subfactor. The major subfactors operating in the forest environment are (1) amount of bare soil, or conversely, ground cover, (2) canopy, (3) soil reconsolidation, (4) soil organic content, (5) fine roots, (6) residual binding effect, (7) on-site storage, (8) steps, and (9) contour tillage. Subfactors 1, 2, 3, 5, 6 and 7 have direct counterparts in agricultural practices. Steps do not occur in most agricultural settings. Contour tillage is the supporting practices factor P.

***Bare Soil Subfactor**: Erosion is a function of the amount of exposed soil. Cover such as litter, slash, logs, and surface rock protects soil from the erosive forces of raindrop impact and runoff. Exposed forest soils are subject to soil detachment by raindrop impact. Also, they yield surface runoff, which potentially erodes soil and transports detached soil from the slope. The observer estimates the amount of bare soil and enters that percentage into a relationship to get a subfactor value for bare soil.*

If the forest floor is completely covered with litter, entering zero percent bare soil into relationship yields a zero for the subfactor value. Protected and undisturbed forest soils have infiltration rates that usually exceed rainfall intensity. Entering a zero into the equation and multiplying it against the other factor and subfactor values will result in a zero erosion estimate. This is consistent with research results and observations made on undisturbed forest. Undisturbed and protected forest soils will infiltrate all but the rarest rainfall intensities, yield no surface runoff, and experience no soil detachment or transport over the soil surface.

Canopy Subfactor: *Vegetal canopy intercepts rainfall and collects water on its foliage. Water drops form and fall to the ground. Drops falling from the canopy may be larger than the original raindrops, but they fall from a low canopy; the energy of the drops reaching the soil surface is less than that of rainfall in open areas. Some of the intercepted rainfall never reaches the ground, but is evaporated during and after the storm. Some of the intercepted rainfall reaches the ground as stemflow and may contribute to runoff. Wischmeier (1975) developed values for the canopy subfactor that depend of foliage density and average drop height. The average drop height is approximately the midpoint in many types of canopy.*

Soil Reconsolidation Subfactor: *Soil reconsolidates and becomes less erodible over time after land is retired from tillage. At Zanesville, Ohio, after 7 years erosion on plots retired from tillage and kept bare reduced to 45 percent of the erosion when they were maintained in tilled, continuous fallow (Wischmeier 1973). The 0.45 value corresponds to the C factor for undisturbed land with no cover (Wischmeier and Smith 1978). This soil-type subfactor is necessary because the soil erodibility factor K is derived from tilled soils in continuous fallow; that is, continuously void of vegetative cover. A relationship for decreasing erodibility over time as soil reconsolidates is presented.*

For untilled forest soils, the soil reconsolidation subfactor is 0.45. However, if the soil is tilled by disking, or rootraking 2 inches or more deep, this subfactor begins at 1.0 and decreases with time after tillage. To evaluate soil reconsolidation, the observer determines whether the soil has been tilled or not, and if tilled, how long ago.

High Organic Content Subfactor: *Under permanent forest, topsoil accumulates a high organic matter content and in some cases it is higher than that used in the USLE soil erodibility nomograph, which only goes as high as 4 percent organic matter. For those forest soils that have more than 4 percent, an adjustment factor of 0.7 is assigned this subfactor.*

Fine Root Subfactor: *A dense mat of fine roots is usually present in the top 2 inches of forest soils. Even after the trees are removed, the residual root mat will partially protect soil from erosive forces of rainfall*

and runoff by holding the soil in place. Sometimes the site is exposed by removal of the surface organic material, while the topsoil with its fine root mat is left in place. Where equipment has removed the topsoil, the fine root mat is usually eliminated. Careful examination is often required to see fine roots. The subfactor is evaluated by estimating the percent of the bare soil with fine roots present and entering into a relationship to get the subfactor value.

Residual Binding Effect Subfactor: *The erosion response of a soil depends on the soil's recent history. That is, there is a residual or carryover effect when the land use or condition changes. When a soil that has not been tilled for some time is cultivated, erosion immediately after it is first tilled may be much less than it will be 2 or 3 years later. At first the soil has a fairly good structure and fine roots and organic matter bind soil into more stable aggregates. With time, this effect decays and the soil becomes more erodible.*

The magnitude of the effect, and its duration, is a function of the amount of roots and organic matter in the soil at the time of tillage, plus structure and permeability of the subsoil.

Onsite Storage Subfactor: *Not all detached soil may be delivered to the toe of the slope; a portion may be stored locally in depressions, behind slash and logs, and in litter on the slope. Values range from 0 to 1.0 for forest conditions. A "0" means all detached soil is stored locally on the slope; none reaching the toe of the slope. A "1.0" means there is no storage on the slope. The observer evaluates onsite storage by estimating the proportion of the existing onsite erosion that will be trapped locally.*

Step Subfactor: *Surface runoff often washes debris down slope until it lodges. This debris accumulates and forms miniature dams which pond water and collect sediment. When these miniature dams are filled with sediment, they form steps. Steps also form behind roots, clumps of vegetation, and other obstacles, and when depressions fill with sediment. On steep slopes, cattle trails on the contour can form steps.*

Steps reduce slope steepness on the area occupied by steps. The step subfactor is evaluated by estimating the percentage of the slope occupied by steps, measuring the slope gradient, and entering these observations into a relationship.

Contour Tillage Subfactor: *Disking on the contour generally reduces sheet and rill erosion by reducing runoff amount and velocity in comparison to tillage up and down slope. Disking equipment should be operated on the contour, but this in not always practical, resulting in ridges and furrows being oriented at an angle to the contour. As furrows and ridges increasingly deviate from the contour, their effectiveness decreases. As the grade along the furrows increases, the amount of material deposited in furrows quickly decreases. The value*

for this subfactor is a function of degrees off contour by the furrows and land slope.

The potential subfactors by disturbance category are presented in Table 6. Not all subfactors will be operating on a site. Therefore, the observer must determine which subfactors are operating and make the necessary observations to develop subfactor values. These subfactor values are multiplied together to product the CP factor for the equation. The product of factor values for R, K, LS and CP equals the estimated erosion rate.

The use of the USLE should be limited to comparing sheet and rill erosion between management practices and the effectiveness of BMP's in minimizing sheet and rill erosion. The best use of the USLE is to estimate long-term average erosion rates for the management condition and use it as a planning tool. The reason for the long term rate is that the R (rainfall) factor is the average rainfall energy computed for precipitation records exceeding 20 years in length.

Rainfall energy can be computed from recording rain gage records for a individual year or a season. Based upon the compute R, erosion can be estimated for a year or season.

Table 6. Potential subfactors by disturbance category

Subfactor	Disturbance Category	
	Tilled	Untilled
Bare soil	X	X
Canopy	X	X
Soil reconsolidation	X	X
Soil organic content	X	
Fine roots	X	X
Residual binding	X	
Onsite storage	X	X
Steps	X	X
Contour tillage	X	

Fabric Dams

Dissmeyer (1982b) developed the use of fabric dams to measure erosion on slopes and demonstrate the effectiveness of BMP's in reducing slope erosion. These have been used in erosion research. Fabric dams can be used in BMP Effectiveness II, III and IV monitoring.

Fabric dams are relatively easy and inexpensive to construct (Figure 10). The materials include a rot-resistant and water-permeable fabric, treated posts, lumber, hog wire, staples, and nails. Posts are set 4 to 8 feet apart 18 to 24 inches into the ground. The posts are braced to prevent the weight of water and sediment from collapsing the dam. Posts should be 3 feet above the ground. A 2 x 4 is nailed across the tops of the posts to form a top rail. The dam should be about 30 feet wide on the contour and have wings at the ends extending straight up-slope to keep runoff and sediment from flowing around the end of the dam. On steep slopes the wings extend up slope to a point equal in elevation to the top rail on the dam. On gentle slopes the wings should extend 10 feet up slope.

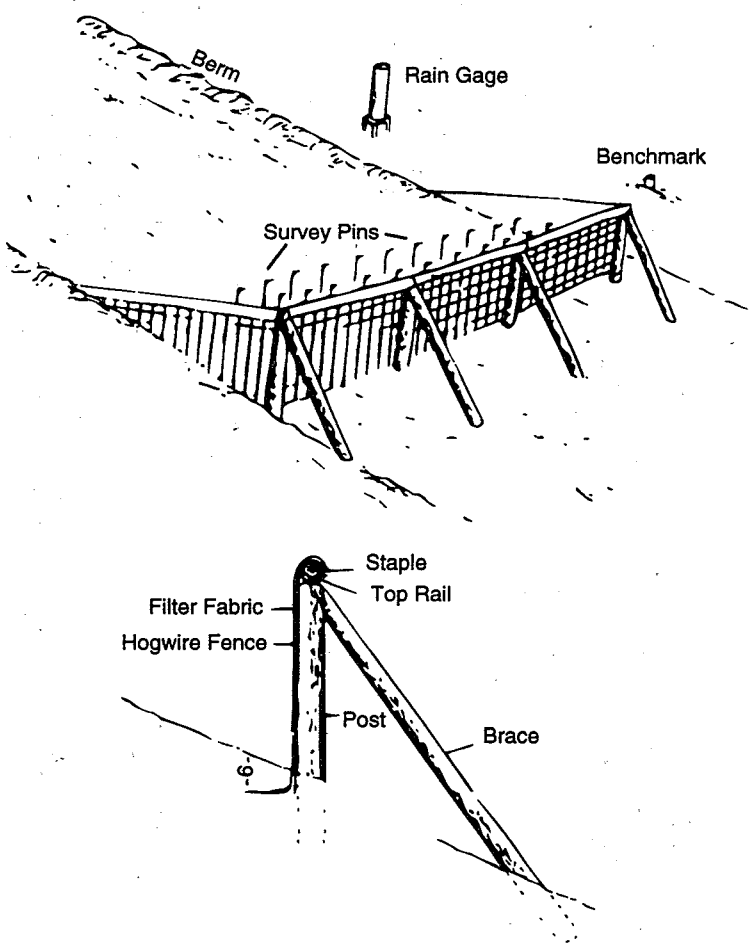

Figure 10. Fabric dam construction.

A 1- by 1-foot grid of flagged pins is installed immediately up-slope of the dam to facilitate measuring the volume of trapped sediment. A bench mark is established near the dam and the ground elevation at each pin in the grid is measured with a rod and level. As sediment accumulates, its elevation at each pin increases. The volume of sediment trapped is determined by measuring the elevation of sediment at each pin, computing the elevation change at each pin, and multiplying the average depth of deposit by the area of sediment deposition on the grid. To translate volume of sediment into weight, the bulk density of the trapped sediment is determined and multiplied by the volume. Sediment can be measured annually, seasonally, or after a major storm.

The plot boundary above the dam needs to be established by building a berm and covering it with litter or by installing plastic garden edging. The recommended size of plot is 70 to 100 feet long by 25 to 35 feet wide on slopes. Smaller dams can be used to measure erosion from a road dip or a waterbar on a skid trail. Fabric dams are not recommended for streams because the dams can be overtopped or stream flow can be diverted around the end, causing stream-bank erosion.

Flumes and Samplers

Flumes and samplers are appropriate for the BMP Effectiveness IV and occasionally at level III. Researchers frequently install flumes and samplers to measure erosion, runoff, and water quality from watershed slopes, small drainages, or specific management areas, such as road drainages, road fills, road banks, etc. For these purposes, the flumes and samplers need to be located on the slope with defined plots or in ephemeral drains. Watershed or plot boundaries are mapped or established to define the area above the flume.

The volume of runoff is measured by stage gages in the flumes or by proportional samplers. The proportional samplers collect a prescribed percent of the water and sediment. The sampled water can be analyzed for nutrients and other constituents.

Pump samplers are often used to collect bottles of water at fixed time intervals during runoff events. Pump samplers can be turned on by a rise of water in the flume so that sampling intensity is increased during rainfall/runoff events. Each water sample collected is stored in a separate bottle, allowing sediment concentrations to be analyzed at various times during the study.

These installations require periodic maintenance and collection of samples for analysis. They are usually accompanied by recording raingages and weather stations to facilitate analysis of runoff/precipitation relationships. Servicing all the equipment is labor intensive and expensive.

Rainfall Simulators

Rainfall simulators have been used for evaluating erosion and refining erosion prediction methods (e.g., USLE). They are research tools to evaluate different management practices and on-ground conditions, thus appropriate for BMP Effectiveness IV assessment. Simulators are quick way to test the effectiveness of BMP's in reducing erosion and surface runoff.

Rainfall simulators take various forms, but have common features. A rainfall simulator applies a known amount of "rain" to a plot of fixed size. The apparatus includes sprinklers at a fixed elevation over the plot, a water source or tank trucks of water, and pumps to transport water from the source to the sprinklers. The sprinkler nozzles can produce droplets of desired sizes to simulate raindrops. The plots will vary in width and length depending on the apparatus used and the conditions being evaluated. The plot boundary is defined by metal or plastic edging. At the base of the plot is a small flume and samplers to measure runoff and sediment.

Herbicide Cards

Cards with surfaces that react to herbicides and are used to monitor drift from aerial applications. Herbicide cards placed at various distances from a stream in buffer strips before spraying reveal if and how far into the buffer drift occurred and if drift deposited herbicides directly into the stream. Herbicide cards can be used at all levels of monitoring. They are used to refine BMP's and to monitor their effectiveness in preventing herbicide drift from reaching the stream.

Herbicide Indicators

For BMP Effectiveness I, empirical observations can determine if the herbicide drift reached a stream. Dead vegetation in the buffer strip reveals how far lethal concentrations drifted. Drift of sublethal concentrations are revealed by close inspection of green leaves, looking for spots and other symptoms of herbicide damage. These observations are made after the herbicide has had time to damage or kill the vegetation.

Chemical and Physical Monitoring Methods

Decades of water-quality research on the impacts of forest management practices have revealed that there are a limited number of variables affected by forest management. MacDonald et al. (1991) present an excellent summary of forest management interactions with 29 of the most commonly affected water-quality variables (see Appendix). The discussions in this section will be limited to 14 chemical and physical variables; most of the remaining 15 are discussed in the biological, habitat, channel geomorphology, and aquatic vegetation sections.

Generally, only a few chemical and physical variables need to be considered in evaluating the effectiveness of forest management practices in meeting water-quality goals or standards (Table 7). Too often, forest monitoring efforts have analyzed water samples for metals and other nonforest-management-related variables. Monitoring such variables should be restricted to other land uses, for example, oil wells, mines, hazardous waste sites, and road construction exposing geologic materials that contribute to pollution or atmospheric deposition. If woodland grazing or developed recreation area occur in the study watershed, fecal coliform would be added to the list of variables measured. A detailed discussion of the significance, measurement, and analytical methods for each of these parameters is presented in MacDonald et al. (1991) in the Appendix.

Table 7. The 14 forest resource management water-quality-related parameters covered in this discussion

Water column	Flow	Sediment
Temperature	Peak flows	Suspended
pH	Low flows	Turbidity
Conductivity	Water yield	Bedload
Dissolved oxygen		
Intergravel DO		
Nitrogen		
Phosphorus		
Herbicides and pesticides		

Chemical and Physical Methods by Monitoring Level

The monitoring approach for each level is discussed in general terms. Not all variables need to be monitored for each level of monitoring. The variables monitored depend on the situation and the issues being addressed.

BMP Effectiveness I

Some empirical evidence can be used to compare the reference stream with the study stream, including: (1) water clarity, color, and odor, (2) presence of scum, (3) turbidity, and (4) brightness of substrate material. The reference and study stream must be observed within a short period of time to make the comparisons valid. The reference and study stream need to be experiencing the same weather at the time of comparison.

Field kits and meters can be used at this level of monitoring. Temperature can be measured by a thermometer. Conductivity, dissolved oxygen, turbidity, and pH are measured by meters in situ. Nitrogen can be analyzed using a field kit. A grab sample can be sent to a laboratory for analysis for suspended sediment and fecal coliform. Given the limited precision of the field kits, this level of monitoring will detect only major differences between reference and study streams.

At this level of monitoring, stream flow comparisons between reference and study streams will be very difficult and unreliable. However, at points with equal watershed area above, high-water marks could be compared. If the reference stream stayed within its banks and the study stream did not, the cumulative effect of activity in the study watershed may have increased peak flows. Judgment on differences in flow can be supported by differences in channel morphology and/or stability.

If management in the study watershed is suspected to have altered low flow, the following procedure can be used. First, locate points on the reference and study streams that are similar in classification and watershed area. Using a current meter in a measured stream profile, measure stream flow during the low-flow period. The flow measurements should be made same day, one immediately after the other. If the flow cannot be measured at points with essentially the same watershed area, converting flow measures to cubic feet per second per square mile will allow comparison.

BMP Effectiveness II

The same basic procedures in Level 1 are applied in BMP Effectiveness II. However, the number of samples and sample points would increase. Samples are collected several times over a period of a few weeks to a couple of months, depending upon the issue and management activity being monitored.

BMP Effectiveness III

Water quality is sampled spatially and temporally in reference and study streams, following a detailed, statistically designed monitoring plan. Depth-integrated sediment sampling, automated pump samplers, recording temperature gages, and similar equipment may be used at this level. Samples are preserved and sent to an analytical lab using USEPA quality assurance/quality control protocols

for analysis. The resulting data are entered and stored in a computer database for management and analysis. Established statistical methods for water-quality and stream-flow analysis are used. This is an appropriate level to monitor herbicides and pesticides because of the importance of timing of sampling with rainstorms and the need for analytical labs to process the samples.

Monitoring ground water quality helps to answer some questions and is appropriate at this level. A couple of examples where ground-water monitoring may be needed are: (1) tracing herbicide movement through the soil to ground water or to a stream, and (2) evaluating the quality of ground water under forested land that was formerly in agriculture to determine if agricultural chemicals are still affecting stream quality.

Raingages need to be located in each of the reference and study watersheds so that precipitation can be measured. Specialized equipment can be installed where atmospheric deposition is an issue. These can be storage gages and/or recording gages, depending upon the issues addressed.

Recording stream gaging stations are often needed to translate concentration data into total yields for a storm, a season, or a year. Stream gages measure stage heights, which can be translated into discharge using rating curves. Long-term flow records can be used to compare peak flows, low flows, and annual water yield between the reference and study streams. It may be necessary to establish recording stream-gaging stations if USGS gaging stations are not available on the reference and study streams.

BMP Effectiveness IV

In addition to the procedures followed in Effectiveness III, five others may be needed:

(1) Automated sampling during the rising and falling stages of runoff for some variables, for example, suspended sediment. Flumes with proportional samplers might be used to sample sediment yields.

(2) Monitoring herbicides and pesticides intensively for an extended period after application to determine the peak concentration, when it occurred, and under what climatic conditions.

(3) Monitoring ground-water quality to trace chemical/nutrient movement through the soil to ground water or to a stream, to measure the peak concentration and determine when it occurred after application, and to determine how long the chemical/nutrient content in soil moisture is elevated.

(4) Rain gage networks, coupled with complete weather stations, should be located in each of the reference and study watersheds. If atmospheric deposition is an issue, it should be sampled and assessed.

(5) Weirs may need to be constructed across streams to increase the accuracy of flow, sediment, nutrient, etc., measurements so watershed budgets can be constructed for each constituent. Long-term, accurate flow records can be used to detect differences in peak flows, low flows, and annual water yield between the reference and study streams.

Channel Geomorphology Monitoring Methods

The geomorphology of stream channels responds to influences of management. Some of these influences are addressed in the discussion of habitat. Here the focus is on channel geomorphologic characteristics: channel cross-sections, channel width/depth ratios, thalweg profile (longitudinal profile of the stream bed), bed material, sinuosity, and bank stability. Another issue addressed is classifying reference and study streams to ensure they are as similar as possible.

Channel Geomorphology Methods by Monitoring Level

The channel geomorphology monitoring methods abstracted below apply to all levels of monitoring. The differences among monitoring levels are merely the degree of implementation. At the BMP Effectiveness I level, empirical comparisons of stability and channel classification are made using knowledge of reference conditions. At the BMP Effectiveness II level, the Pfankuch stability rating is used, along with a detailed Rosgen's channel classification of reference and study streams. At the BMP Effectiveness III and IV levels, the Pfankuch method is appropriate for stability rating analysis between similar reference and study streams. A detailed Rosgen channel classification and mapping, the development and use of bankfull width, average bankfull depth and cross-sectional area to watershed area relationships, and particle size distributions for reference stream comparisons to study streams are appropriate. The V* Pool Sediment or Rosgen Channel Monitoring are appropriate at levels III and IV.

The various methods are listed below and discussed in detail as indicated.

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Empirical/Perceptual	82
Pfankuch Channel Stability	83
Rosgen's Stream Channel Classification, plus channel geomorphologic relationships	95
Pebble Counts	112
V* Pool Sediment	107
Rosgen Channel Monitoring	108
Alaska Channel Classification	105

Empirical/Perceptual

This method is used only for BMP Effectiveness I. The trained observer must be knowledgeable in stream-channel classification and in characteristics of stable and unstable channels. The observer must be also trained in Rosgen's (1985) stream-channel classification system and/or Pfankuch (1978) channel-stability-evaluation methods. He/she should have a mental picture of the channel classes and stability of the reference streams for comparison with the study streams.

Reference and study streams should be as similar as possible, including channel classification and stability. As a study stream passes a management area, there may be a significant change in channel class or stability that is clearly inconsistent with the reference stream. For example, the upstream reach of the study stream may have a stable, single channel. As the stream passes the management area, it may become a multi-channel, braided stream. The observer notes that the bank stabilizing vegetation has been removed by management, resulting in an unstable, braided channel that is atypical of the reference stream.

Streambed material is used in classifying stream channels. The observer may notice a major shift in streambed material (for example, cobble to sand) as the stream passes a road or a management area. If the change is inconsistent with the reference stream conditions and sediment loading can be traced to the management condition, then channel geomorphological changes can be associated with management.

The observer visually compares bank stability, width/depth ratios, streambed stability, aggradation, and degradation between reference and study streams.

Pfankuch Channel Stability

Pfankuch (1978) developed a qualitative method for evaluating the stability of stream channels that can be used at all levels of monitoring. The method evaluates 15 factors and rates their quality based upon narrative descriptions. Depending upon which description best applies to the condition found, a score for that factor is given. The scores for all factors are totalled and the total score is used to rate stability as excellent, good, fair, or poor.

The Pfankuch (1978) method uses a field form containing the following factors, narrative descriptions, and rating scores:

I. Upper Channel Banks

The land area immediately adjacent to the stream channel is normally and typically a terrestrial environment. Landforms vary from wide, flat, alluvial flood plains to the narrow, steep termini of mountain slopes. Intermittently this dry land flood plain becomes a part of the water course. Forces of velocity and turbulence tear at the vegetation and land. These hydraulic forces, while relatively short lived, have great potential for producing on-site enlargements of the stream channel and downstream sedimentation damage. Resistance of the component elements on and in the bank are highly variable. This section is designed to aid in rating this relative resistance to detachment and transport by floods.

1. ***Landform slope:*** *The steepness of the land adjacent to the stream channel determines the lateral extent and ease to which banks can be eroded and the potential volume of slough which can enter the water. All other factors being equal, the steeper the land adjacent to the stream, the greater the potential volume of slough materials.*

   A. *Excellent: Side slopes to the channel are generally less than 30 percent on both banks. (2 points)*

B. *Good: Side slopes up to 40% on one or occasionally both banks. (4 points)*

C. *Fair: Side slopes 40 to 60% common on one or both banks. (6 points)*

D. *Poor: Steep slopes over 60%. (8 points)*

2. ***Mass Wasting Hazard:*** *This rating involves existing or potential detachment from the soil mantle and downslope movement into waterways of relatively large masses of soil mantle. Mass movement of banks by slumping or sliding introduces large volumes of soil and debris into the channel suddenly, causing constrictions or complete damming followed by increased stream flow velocities, cutting power and sedimentation rates. Conditions deteriorate in this factor with proximity, frequency and size of the mass wasting areas and with progressively poorer internal drainage and steeper terrain.*

   A. *Excellent: There is no evidence of mass wasting that has or could reach the stream channel. (3 points)*

   B. *Good: There is evidence of infrequent and/or very small slumps. Those that exist may occasionally be "raw," but predominately the areas are revegetated and relatively stable. (6 points)*

   C. *Fair: Frequency and/or magnitude of the mass wasting situation increases to the point where normal high water aggravates the problem of channel changes and subsequent undercutting of unstable areas with increased sedimentation. (9 points)*

   D. *Poor: Mass wasting is not difficult to detect because of the frequency and/or size of existing problem areas or the proximity of banks are so close to potential slides that any increases in the flow would cut the toe and may trigger slides of significant size to cause downstream water quality problems for many years. (12 points)*

3. ***Debris Jam Potential:*** *Floatable objects are deposited on stream banks by man and as a natural process. By far, the bulk of this debris in natural in origin. Tree trunks, limbs, twigs, and leaves reaching the bulk of the obstructions, flow deflectors, and sediment traps are rated. This factor assesses the potential for increasing these impediments to the natural direction and force of flow where they now lay. Also, it includes the possibility of creating new debris jams under certain flow conditions.*

   Caution: Woody debris provides vital habitat for fish and benthic macroinvertebrates and is part of productive natural streams. Because woody debris is associated with some stability problems does not mean it has to be removed.

A. *Excellent: Debris may be present on the banks, but is so situated or is of such size, that the stream is not able to push or float it into the channel and, therefore, for all intents and purposes, it is absent. In truth, there may be none physically present. Both situations are rated the same. (2 points)*

B. *Good: The debris present offers some bank protection for a while but is small enough to be floated away in time. Only small jams could be formed with this material alone. (4 points)*

C. *Fair: There is a noticeable accumulation of all sizes and the stream is large enough to float it away, at certain times, thus decreasing the bank protection and adding to the debris jam potential downstream. (6 points)*

D. *Poor: Moderate to heavy accumulations are present due to fires, insect attack, disease mortality, windthrow, or logging slash. High flows will float some debris away and the remainder will cause channel changes. (8 points)*

4. ***Vegetative Bank Protection:*** *The soil in banks is held in place largely by plant roots. Riparian plants have almost unlimited water for both crown and root development. Their root mats generally increase in density with proximity to the open channel. Trees and shrubs generally have deeper root systems than grasses and forbs. Roots seldom extend far into the water table, however, and near the shore of lakes and streams they may be compartively shallow rooted. Some species are, therefore, subject to windthrow.*

In addition to the benefits of the root mat in stabilizing the banks, the stems help to reduce the velocity of flood flows. Turbulence is generated by stems in what may have been laminar flow. The seriousness of this energy release depends on the density of both overstory and understory vegetation. The greater the density of both, the more resistance displayed. Damage from turbulence is greatest at the bank edge and diminishes with distance from the normal channel. Other factors to consider, in addition to the density of stems, are the varieties of vegetation, the vigor of growth, and the reproduction processes. Vegetal variety is more desirable than a monotypic plant community. Young plants, growing and reproducing vigorously, are better than old, decadent stands.

A. *Excellent: Trees, shrubs, grass and forbs combined cover more than 90 percent of the ground. Openings in this nearly complete cover are small and evenly dispersed. A variety of species and age classes are represented. Growth is vigorous and reproduction of species in both the under- and over-story is proceeding at a rate to insure continued ground cover conditions. A deep, dense root mat is inferred. (3 points)*

B. *Good: Plants cover 70 to 90 percent of the ground. Shrub species are more prevalent than trees. Openings in the tree canopy are larger than the space resulting from the loss of a single mature individual. While the growth vigor is generally good for all species, advanced reproduction may be sparse or lacking entirely. A deep root mat is not continuous, and more serious erosive incursions are possible in the openings. (6 points)*

C. *Fair: Plant cover ranges from 50 to 70 percent. Lack of vigor is evident in some individuals and/or species. Seedling reproduction is nil. This condition is ranked fair, based primarily on the percent of the area not covered by vegetation with a deep root mat potential and less on the kind of plants that make up the overstory. (9 points)*

D. *Poor: Less than 50 percent of the ground is covered. Trees are essentially absent. Shrubs largely exist in scattered clumps. Growth and reproduction vigor is generally poor. Root mats discontinuous and shallow. (12 points)*

II. Lower Channel Banks

The channel zone is located between the normal high water and low water lines. Both aquatic and terrestrial plants may grow here but normally their density is sparse.

The lower channel banks define the present stream width. Stability of these channel banks is indicated under a given flow regimen by minor and almost imperceptible changes in channel width from year to year. In other words, encroachment of the water environment into the land environment is nil. Under conditions of increasing channel flow, the banks may weaken and both cutting (bank encroachment) and deposition (bank extension) begin, usually at bends and points of constriction. Cutting is evidenced by steepening of the lower banks. Eventually the banks are undercut, followed by cracking and slumping. Deposition behind rocks or bank protrusions increase in length and depth.

As the channel is widened, it may also be deepened to accommodate the increased volume of flow. For convenience only, changes of channel bottoms are observed separately and last in this evaluation scheme.

5. ***Channel Capacity:*** *Channel width, depth, gradient, and roughness determine the volume of water which can be transmitted. Over time, channel capacity has adjusted to the size of watershed above the reach rated, to climate, and to changes of vegetation. Some indicators of change are widening and/or deepening of the channel, which affects the ratio of width to depth. When the capacity is exceeded, deposits of soil are found on the backs and organic debris may be found hung up in the bank vegetation. These are expressions of the most recent flood event. Indicators of conditions as recent as a year or two ago may be difficult or impossible to find, but do your best to estimate what normal*

peak flows are and whether the present cross section is adequate to handle the flow without bank deterioration.

A. *Excellent: Cross sectional area is ample for present peak volumes plus some additional, if needed. Over-bank floods are rare. Width to depth ratio less than 7. (1 point)*

B. *Good: Adequate cross sectional areas contain most peak flows. Width to depth ratio 8 to 15. (2 points)*

C. *Fair: Channel barely contains the peak runoff in average years or less. Width to depth ratios range from 15 to 25. (3 points)*

D. *Poor: Channel capacity generally inadequate. Overbank floods quite common as indicated by kind and conditions of bank plants and the position and accumulation of debris. Width to depth ratio 25 or more. (4 points)*

6. ***Bank Rock Content:*** *Examination of the materials that make up the channel bank will reveal the relative resistance of this component to detachment by flow forces. Since the banks are perennially and intermittently both aquatic and terrestrial environments, these sites are harsh for most plants that make up both types. Vegetation is therefore generally lacking, and it is the volume, size, and shape of the rock component that primarily determine the resistance to flow forces.*

   A soil pit need not be dug. Surface rock and exposed cut banks will enable categorization of this item as percentage ranges.

   A. *Excellent: Rock makes up 65% or more of the volume of the banks. Within this rock matrix large angular boulders 12+ inches (on their largest axis) are numerous. (2 points)*

   B. *Good: Banks 40-65% rock that are mostly small boulders and cobble ranging in size from 6-12" mean diameter. Some may be rounded while others are angular. (4 points)*

   C. *Fair: 20-40 of bank volume rock. While some big rock may be present, most fall into the 3-6" diameter class. (6 points)*

   D. *Poor: Less than 20% rock fragments, mostly of gravel sizes 1-3" in diameter. (8 points)*

7. ***Obstructions and Flow Deflectors:*** *Objects within the stream channel, like large rocks, embedded logs, bridge pilings, etc., change the direction of flow and sometimes the velocity as well. Obstructions may produce adverse stability effects when they increase the velocity and deflect the flow into unstable and unprotected banks and across unstable bottom materials. They also may produce favorable impacts when velocity is decreased by turbulence and pools are formed.*

Sediment Traps: *Channel obstructions that dam the flow partly or wholly form pools or slack water areas. The pools lower the channel gradient. With this loss of energy, the sediment transport power is greatly reduced. Coarse particles drop out first at the head of the pool. Some or all the fine suspended particles may carry on through.*

Embedded logs and large boulders can produce very stable natural dams that do not add to channel instability. Some debris dams and beaver dams, however, are quite unstable and only serve to increase the severity of channel damage when they break up.

The effectiveness of these sediment traps depends on pool length relative to entrance velocity. The swifter the current, the longer the pool needed to reach zero velocity. Turbulence caused by falls at the head of the pool shortens the length required to reach zero velocity.

How long these traps are effective depends on depth and width as well as pool length and, of course, the rate of sediment accretion.

Types of vegetation growing in the water, like alders, willows, cattails, reeds, and sedges, are also effective traps in some locations and reduce flow velocity and sediment carrying power.

A. *Excellent: Logs, rocks, and other obstructions to flow are firmly embedded and produce a pattern of flow that does not erode the banks and bottom or cause sediment buildups. Pool riffle relationship stable. (2 points)*

B. *Good: Obstructions to flow and sediment traps are present, causing cross currents that create some minor bank and bottom erosion. Some of the obstructions are newer, are not firmly embedded, and move to new locations during high flows. Some sediment is trapped in pools, decreasing their capacity. (4 points)*

C. *Fair: Moderately frequent and quite often unstable obstructions cause noticeable seasonal erosion of the channel. Considerable sediment accumulates behind obstructions. (6 points)*

D. *Poor: Obstructions and traps are so frequent they overlap, are often unstable to movement, and cause a continual shift of sediments at all seasons. Since traps are filled as soon as formed, the channel migrates and widens. (8 points)*

Cutting and Deposition: *are processes that occur at the same time, but at different locations along a stream. You can not have one without the other. However, it is possible for each to be taking place in different reaches of the same stream at the same time, and hence the separation for classification purposes.*

8. ***Cutting:*** *One of the first signs of channel degradation is a loss of aquatic vegetation by scouring or uprooting. Some channels are naturally devoid of aquatic plants, and here the first stages would be an increase in the steepness of the channel banks. Beginning near the top, and later extending in serious cases to the total depth, the lower channel bank becomes a near vertical wall.*

   If plant roots bind the surface horizon of the adjacent upper bank into a cohesive mass, undercutting will follow. This process continues until the weight of overhang causes the sod to crack and subsequently slump into the channel. Differential horizontal compaction and texture could also result in undercut banks even with an absence of vegetative cover. There are some loosely consolidated banks that with or without vegetation are literally nibbled away, never developing much, if any, overhang.

   A. *Excellent: Very little or no cutting is evident. Raw, eroding banks are infrequent, short, and predominately less than 6" high. (4 points)*

   B. *Good: Some intermittent cutting along channel outcurves and at prominent constrictions. Eroded areas are equivalent in length to one channel width or less, and the vertical cuts are predominately less than 12". (8 points)*

   C. *Fair: Significant bank cutting occurs frequently in the reach. Raw vertical banks 12" to 24" high are prevalent as are root mat overhangs and sloughing. (12 points)*

   D. *Poor: Nearly continuous bank cutting. Some reaches have vertical cut faces over 2 feet high. Undercutting, sod-root overhangs, and vertical side failures may also be frequent in the rated reach . (16 points)*

9. ***Deposition:*** *Lower bank channel areas are generally the steeper portions of the wetted perimeter and may be rather narrow strips of land that offer slight opportunity for deposition. Exceptions to this abound, because deposition is often noted on the lee side of large rocks and log deflectors that form natural jetties. However, these deposits tend to be short and narrow. On the less steep, lower banks, deposition during recession from peak flows can be quite large. The appearance of sand and gravel bars where they did not previously exist may be one of the first signs of upstream erosion. These bars tend to grow, primarily in depth and length, with continued watershed disturbance(s). Width changes are in a shoreward direction as overflow deposition takes place on the upper banks. Dimensional deposition "growth" is limited by the size and orientation of the obstructions to flow along the channel banks, flow velocity, and a continuing upstream sediment supply.*

Deposition may also occur on the inside radii of bends, particularly if active cutting is taking place on the opposite shore. Also, deposits are found below constrictions or where there is a sudden flattening of stream gradient as occurs upstream of geologic nick points.

A. *Excellent: Very little or no deposition of fresh silt, sand, or gravel in channel bars in straight reaches or point bars on the inside banks of curved reaches. (4 points)*

B. *Good: Some fresh deposits on bars and behind obstructions. Sizes tend to be predominately from the larger size classes—coarse gravels. (8 points)*

C. *Fair: Deposits of fresh, coarse sands and gravels observed with moderate frequency. Bars are enlarging and pools are filling so riffle areas predominate. (12 points)*

D. *Poor: Extensive deposits of predominately fresh, fine sands, some silts, and small gravels. Accelerated bar development common. Storage areas are now full and sediments are moving even during low flow periods. (16 points)*

III. Channel Bottom

Water flows over the channel bottom nearly all of the time in perennial streams. It is, therefore, almost totally an aquatic environment, composed of inorganic rock constituents found in an infinite variety of kinds, shapes, and sizes. It is also a complex biological community of plant and animal life.

Both components, by their appearance alone and in combination, offer clues to the stability of the stream bottom. They are arbitrarily separated and individually rated for convenience and emphasis during the evaluation process. Because of the high reliance on the visual sense, inventory work is best accomplished during the low flow season and when the water is free of suspended or dissolved substances.

10. ***Rock Angularity**: Rocks from stratified, metamorphic formations break out and work their way into channels as angular fragments that resist tumbling. Their sharp corners and edges wear and are rounded in time, but they resist the tumbling motion. These angular rocks pack together well and may orient themselves like shingles (imbricated). In this configuration they are resistant to detachment.*

    In contrast, igneous rocks often produce fragments that round up quickly, pack poorly, and are easily detached and moved downstream.

    Excellent to poor ratings relate to the amount of rounding exhibited and, secondarily, the smoothness or polish the surfaces have achieved. Some rocks never do smooth up in the natural environment,

but most round up in time. Both conditions, of course, are relative within the inherent capability of the respective rock types.

A. *Excellent: Sharp edges and corners, plane surfaces roughened. (1 point)*

B. *Good: Rounded corners and edges, surfaces smooth and flat. (2 points)*

C. *Fair: Corners and edges well rounded in two dimensions. (3 points)*

D. *Poor: Well rounded in all dimensions, surfaces smooth. (4 points)*

11. ***Brightness:*** *Rocks in motion "gather no moss", algae, or stain. They become polished by frequent tumbling and, as a general rule, appear brighter in their chroma values than similar rocks that have remained stationary. The degree of staining and vegetative growths relate also to water temperature, seasons, nutrient levels, etc. In some areas a "bright" rock will be "dulled" in a matter of weeks or months. In another it may take years to achieve the same results. Nevertheless, even slight changes during the spring runoff should be detectable during the next summer's survey. Look for changes in the sands and gravels.*

    A. *Excellent: Less than 5% of the total bottom should be bright, newly polished and exposed surfaces. Most will be covered by growths or a film of organic stain. Stains may also be from minerals dissolved in the water. (1 point)*

    B. *Good: 5 to 35% of the bottom appears brighter, some of which may be on the larger rock sizes. (2 points)*

    C. *Fair: About a 50-50 mixture of bright and dull with a 15% leeway in either direction (i.e., a range of 35 to 65% bright materials). (3 points)*

    D. *Poor: Bright, freshly exposed rock surfaces predominate with more than 65% bright materials. (4 points)*

12. ***Consolidation (Particle Packing):*** *Under stable conditions, the array of rock and soil particle sizes pack together. Voids are filled. Larger components tend to overlap like shingles (imbricate). So arranged, the bottom is quite resistant to even exceptional flow forces. Some rock types (granitics) are less amenable to this packing process and never reach the stable state of other like Belt Series rocks.*

A. *Excellent: An array of sizes are tightly packed and wedged with much overlapping which makes it difficult to dislodge by kicking. (2 points)*

B. *Good: Moderately tight packing of particles with fast water parts of the cross section protected by overlapping rocks. These might be dislodged by higher than average flow conditions. (4 points)*

C. *Fair: Moderately loose without any pattern of overlapping. Most elements might be moved by average high flow conditions. (6 points)*

D. *Poor: Rocks in loose array, moved easily by less than high flow conditions and move underfoot while walking across the bottom. The shape of these rocks tends to be predominantly round and sorted so that most are of similar size. (8 points)*

13. ***Bottom Size Distribution and Percent Stable Materials:*** *Rocks remaining on a stream's bottom reflect the geologic sources within the basin and the flow forces of the past. Normally, there is an array of sizes expected to be seen in a given local. After a little experience, observer will be sensitive to abnormal situations. For example, in the mature topography typical of the mountainous West, the flow in small, steep streams is sufficient to wash fine sediments and gravels away. What remains is a gravelly, cobble stream bottom. In the lower reaches where the gradient is less and flow is often slower, deposition of the "fines" eroded above begin to drop out. The separates of sand, silt, and some clay begin to cover the coarser elements. Except where trapped in still water areas, these fines tend to be in constant motion to ever lower elevations.*

Two elements of bottom stability are rated in this item: (1) Changes or shifts from the natural variation of component size classes and (2) the percentage of all components that are judged to be stable materials. Bedrock, large boulders, and cobblestones ranging in size from one to three feet or more in diameter are considered "stable" elements in the average situation. Obviously, smaller rocks in smaller channels might also be classed as stable. The sizes are given only to guide thought. Bedrock as a major component of bottom and banks, no matter what size the channel or how the other elements rate, always results in an excellent classification of that reach.

A. *Excellent: There is no noticeable change in size distribution. The rock mixture appears to be normal for the kind of geologic sources in the basin and the flow forces of streams of this size and location in the watershed.*

If a shift or change has taken place so there are greater percentages of large rock in the small streams and smaller sizes in large streams, the condition class most appropriate should be checked. It is a matter of degree as follows:

(Stable materials 80-100%) (4 points)

B. *Good: Slight shift in either direction. Stable materials 5-80%. (8 points)*

C. *Fair: Moderate shift in size classes. Stable materials 20-50%. (12 points)*

D. *Poor: Marked, a pronounced shift. Stable materials less than 20%. (16 points)*

14. ***Scouring and/or Deposition:*** *Items of size, angularity, and brightness rated above leads to some conclusions as to the amount of scouring and/or deposition that is taking place along the channel bottom.*

A. *Excellent: Neither scouring nor deposition is much in evidence. Up to 5% of either or a combination of both may be present along the length of the reach. (6 points)*

B. *Good: Affected length ranges from 5 to 30%. Cuts are found mostly at channel constrictions or where the gradient steepens. Deposition is in pools and backwater areas. Sediment in pools tends to move on through so pools change only slightly in depth but greatly in composition of their size classes. (12 points)*

C. *Fair: Moderate changes are occurring. 30 to 50% of the bottom is in a state of flux. Cutting is taking place below obstructions, at constrictions and on steep grades. Deposits in pools now tend to fill the pool and decrease their size. (18 points)*

D. *Poor: Both cutting and deposition are common; 50% plus of the bottom is moving not only during high flow periods but at most seasons of the year. (24 points)*

15. ***Aquatic Vegetation:*** *When some measure of stabilization of the soil-rock components is achieved, the channel bottom becomes fit habitat for plant and animal life. This process begins in the slack water areas and eventually may include the swift water portions of the stream cross section. With a change in volume of flow and/or sedimentation rates, there may also be a temporary loss of the living elements in the aquatic environment. This last item attempts to assess the one macro-aquatic biomass indicator found to best express a change in channel stability.*

Clinging Moss and Algae: *These lower plant forms do not have roots but cling to the substrate. They are low growing and may first appear as a green to yellow-green slick spot on the bottom rocks. Moss plants continue with slight variation in color but no great change in mass from season to season. Algae by contrast have a peak of growth activity and then die off in great numbers. The slippery conditions they produce persists after death, however.*

Both algae and moss inhabit the swift water areas as well as the quiet pools and backwater portions of the stream bottom.

A. *Excellent: Clinging plants are abundant throughout the reach from bank to bank. A continuous mat of vegetation is not required but moss and/or algae are readily seen in all directions across the stream. (1 point)*

B. *Good: Plants are quite common in the slower portions of the reach but thin out or are absent in the swift flowing portions of the stream. (2 points)*

C. *Fair: Plants are found but their occurrence is spotty. They are almost totally absent from rocks in swifter portions of the reach and may also be absent in some of the slow and still water areas. (3 points)*

D. *Poor: Clinging plants are rarely found anywhere in the reach. (This is an unusual situation, but it could happen under a combination of adverse environmental conditions). (4 points)*

The rating scores for all elements are added to get a total, stability rating score. Based upon the total score, the stability of the reach is rated: <38 = excellent, 39-76 = good, 77-114 = fair, and 115+ = poor.

Pfankuch's method can be used to compare study and reference stream stability. Again, it is *critical* that the reference and study streams have the same stream channel classification for valid comparisons of channel stability. Rosgen's (1993) channel classification system stratifies streams by geomorphologic characteristics, which translates into an evaluation of relative stability. The degree of channel stability varies by channel type, i.e., the stage of geomorphologic development of the channel and land forms. The best channel stability rating for an unstable stream class may be 85; for a stable stream class it may be 50. Comparing different channel classes leads to erroneous judgments of management impacts on channel stability.

If there is a significant difference in the total score between study and reference streams, comparing individual element scores should reveal what elements are different. The next step is to determine if an impaired element in the study

stream can be linked to a management activity or some other cause in the watershed.

Rosgen's Stream Channel Classification

Rosgen (1985) developed a stream classification system, which is primarily used for classifying reference and study streams. However, it can be used to monitor changes in channel geomorphology. The narrative below comes from Rosgen's 1985 paper; the description of the stream classes reflect 1993 revisions (personal communication). This method can be used at all levels of monitoring, but the level of detail will vary.

> *It has long been a goal for individuals working with rivers to define and understand the processes which influence the pattern and character of river systems. Their differences as well as their similarities under diverse setting pose a real challenge for study. One consistent axiom associated with rivers is that what initially appears complex is even more so under further investigation.*
>
> *Obviously, over-simplification of such complex systems may appear presumptuous. However, the need to categorize stream and river systems by channel morphology is apparent for the following reasons: (1) the need to predict a river's behavior from its appearance; (2) the need to extrapolate specific data collected on a given river reach to another of similar character and; (3) the need to provide a consistent and reproducible frame of reference for those working with river systems.*
>
> ***Stream Classification Criteria***
> *The purpose of this classification scheme is to categorize natural stream channels on the basis of measurable morphological features. Thus, consistent and reproducible descriptions and interpretations can be readily obtained over a wide range of hydrophysiographic regimes.*
>
> *There are many observable stream channel features governed by the laws of physics which operate to form the morphology of the present day channel. Stream morphology and related channel patters are directly influenced by eight major variables including width, depth, velocity, discharge, slope, roughness of channel materials, sediment load and sediment size (Leopold et al., 1964). A change in any one of these variables sets up a series of concurrent changes in the others, resulting in altered channel patterns. Since stream morphology is a result of an integrative process of mutually adjusting variables, those most directly measurable have been incorporated into the delineative criteria for stream types. Selection of the delineative criteria for stream classification was developed from detailed analysis of hundreds of streams over many hydrophysiographic regions and from portions of existing classification schemes.*
>
> *The stream type classification is summarized below, displayed in figures, and includes the following criteria:*

1. *Channel gradient (measured as energy slope of the water surface),*
2. *Sinuosity (ratio of channel length to valley length),*
3. *Width/depth ratio (width at bankful stage divided by bankful depth),*
4. *Dominant particle size of bed and bank materials,*
5. *Entrenchment of channel and confinement of channel in valley (ratio of flood prone width divided by bankful width),*
6. *Landform features, soil erodibility and stability.*

Stream Classification Key

Single Thread Channels	***Channel Class***
- Well Entrenched (Ratio <1.4)	
- Low Width/Depth Ratio (<12)	
- Low sinuosity (<1.2)	
- Slope Range >.10 (Figure 11)	
Channel Materials	
- Bedrock	*A1a+*
- Boulders	*A2a+*
- Cobble	*A3a+*
- Gravel	*A4a+*
- Sand	*A5a+*
- Silt/Clay	*A6a+*
- Slope Range	*.04-.099*
Channel Materials	
- Bedrock	*A1*
- Boulders	*A2*
- Cobble	*A3*
- Gravel	*A4*
- Sand	*A5*
- Silt/Clay	*A6*
- Slope Range .02-.039	
Channel Materials	
- Bedrock	*A1b*
- Boulders	*A2b*
- Cobble	*A3b*
- Gravel	*A4b*
- Sand	*A5b*
- Silt/Clay	*A6b*
Moderate Sinuosity (>1.2)	
- Slope Range .02-.039 (Figure 12)	
Channel Materials	
- Bedrock	*G1*
- Boulders	*G2*
- Cobble	*G3*
- Gravel	*G4*
- Sand	*G5*
- Silt/Clay	*G6*

- *Slope Range <.02*
  - *Channel Materials*
    - *Bedrock* — *G1c*
    - *Boulders* — *G2c*
    - *Cobble* — *G3c*
    - *Gravel* — *G4c*
    - *Sand* — *G5c*
    - *Silt/Clay* — *G6c*

- *Moderate-High Width/Depth Ratio (>12)*
  - *High Sinuosity (>1.4)*
    - *Slope Range .02-.039 (Figure 13)*
      - *Channel Materials*
        - *Bedrock* — *F1b*
        - *Boulders* — *F2b*
        - *Cobble* — *F3b*
        - *Gravel* — *F4b*
        - *Sand* — *F5b*
        - *Silt/Clay* — *F6b*

    - *Slope Range <.02*
      - *Channel Material*
        - *Bedrock* — *F1*
        - *Boulders* — *F2*
        - *Cobble* — *F3*
        - *Gravel* — *F4*
        - *Sand* — *F5*
        - *Silt/Clay* — *F6*

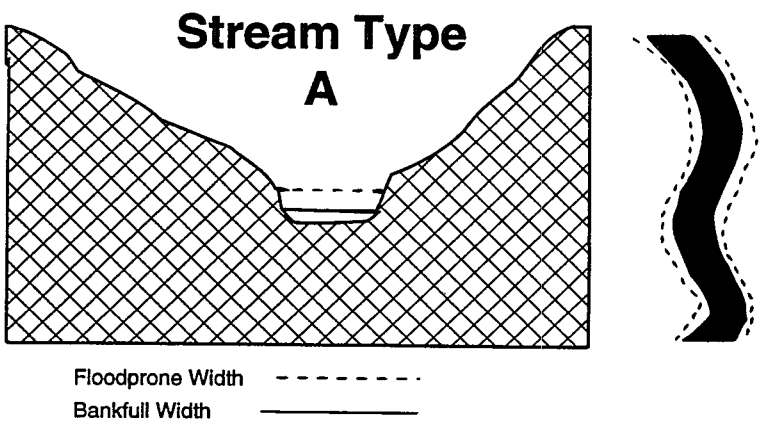

Figure 11. Rosgen Class A Streams.

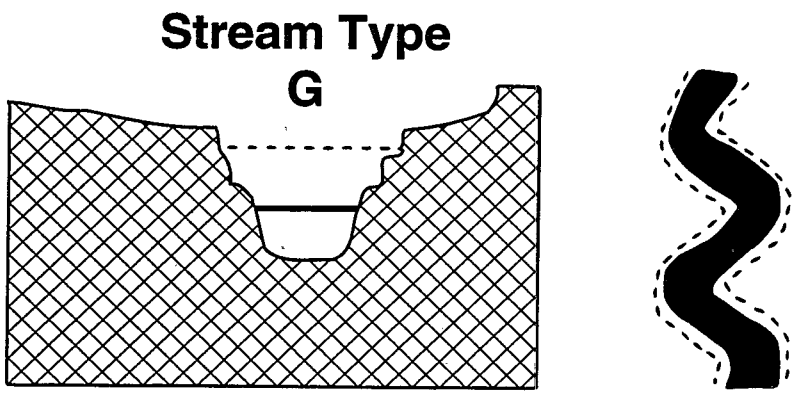

Figure 12. Rosgen Class G Streams.

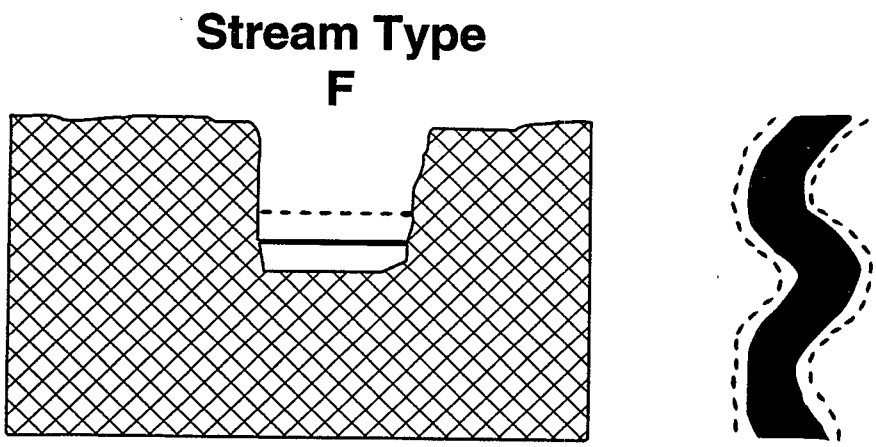

Figure 13. Rosgen Class F Streams.

- Moderately Entrenched (Ratio 1.4-2.2)
  - Moderate Width/Depth Ratio (>12)
    - Moderate Sinuosity (>1.2)
      - Slope Range (.04-.099) (Figure 14)

Channel Material	
- Bedrock	B1a
- Boulders	B2a
- Cobble	B3a
- Gravel	B4a
- Sand	B5a
- Silt/Clay	B6a

      - Slope Range (.02-.039)

Channel Material	
- Bedrock	B1
- Boulders	B2
- Cobble	B3
- Gravel	B4
- Sand	B5
- Silt/Clay	B6

      - Slope Range (<.02)

Channel Material	
- Bedrock	B1c
- Boulders	B2c
- Cobble	B3c
- Gravel	B4c
- Sand	B5c
- Silt/Clay	B6c

- Not Entrenched (Ratio >2.2)
  - Very Low Width/Depth Ratio (<12)
    - Very High Sinuosity (>1.5)
      - Slope Range (.02-.039) (Figure 15)

Channel Material	
- Cobble	E3b
- Gravel	E4b
- Sand	E5b
- Silt/Clay	E6b

      - Slope Range (<.02)

Channel Material	
- Cobble	E3
- Gravel	E4
- Sand	E5
- Silt/Clay	E6

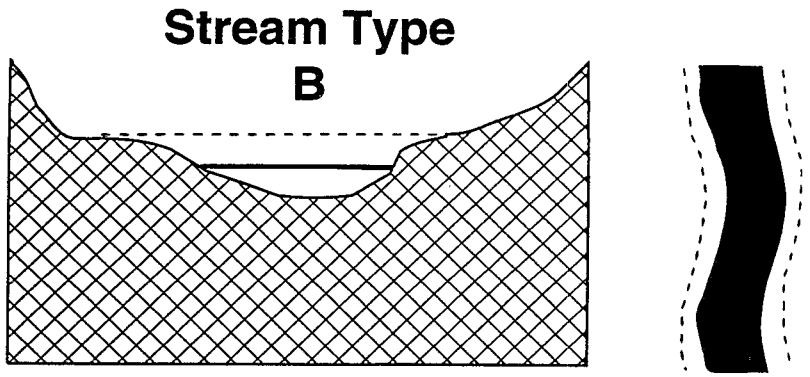

Figure 14. Rosgen Class B Streams.

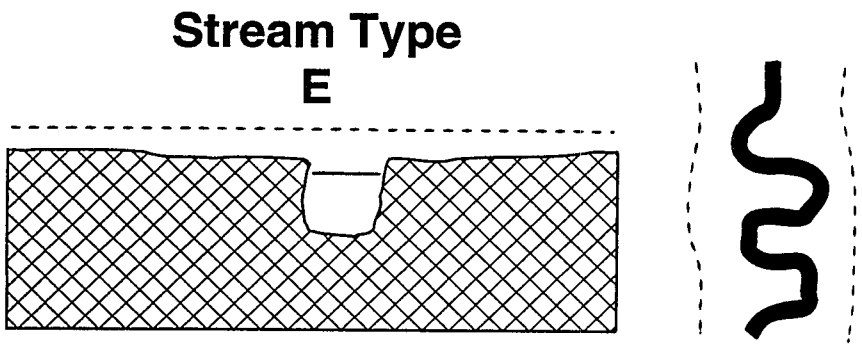

Figure 15. Rosgen Class E Streams.

- *Moderate-High Width/Depth Ratio (>12)*
  - *High Sinuosity (>1.4)*
    - *Slope Range (.02 - .039) (Figure 16)*
      Channel Material

- *Bedrock*	*C1b*
- *Boulders*	*C2b*
- *Cobble*	*C3b*
- *Gravel*	*C4b*
- *Sand*	*C5b*
- *Silt/Clay*	*C6b*

    - *Slope Range (<.02)*
      Channel Material

- *Bedrock*	*C1*
- *Boulders*	*C2*
- *Cobble*	*C3*
- *Gravel*	*C4*
- *Sand*	*C5*
- *Silt/Clay*	*C6*

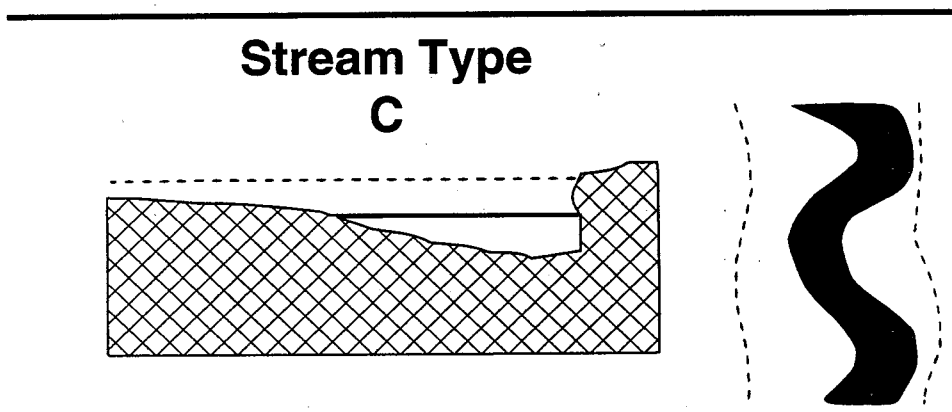

Figure 16. Rosgen Class C Streams.

Multiple Channels

- *Very High Width/Depth Ratio (>50)*

  - *Unstable*

    - *Slope Range (.02-.039) (Figure 17)*
      Channel Material
      - *Cobble* — *D3b*
      - *Gravel* — *D4b*
      - *Sand* — *D5b*
      - *Silt/Clay* — *D6b*

    - *Slope Range (<.02)*
      Channel Material
      - *Cobble* — *D3*
      - *Gravel* — *D4*
      - *Sand* — *D5*
      - *Silt/Clay* — *D6*

  - *Stable*

    - *Slope Range (<.005)*
      Channel Material
      - *Gravel* — *DA4*
      - *Sand* — *DA5*
      - *Silt/Clay* — *DA6*

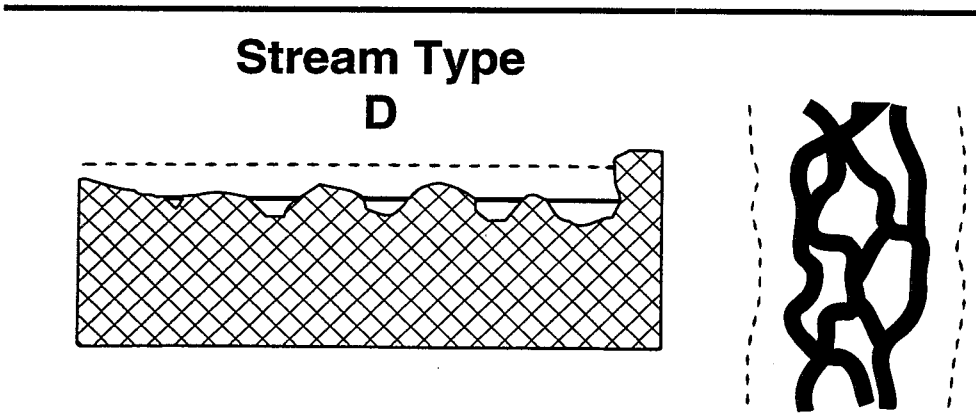

Figure 17. Rosgen Class D Streams.

Bankfull channel width, average channel depth, and channel cross-sectional area can be compared for reference and study streams with the same channel classification. The following discussion is based upon material presented by D. Rosgen and L. B. Leopold in their Short Course on Rivers and Operational Hydrology (1991):

The bankfull discharge is one of the most important characteristics of a channel location. It is not only the discharge that carries over a long time the maximum amount of sediment, but is it the channel forming (channel maintaining) discharge. It is a quantity that integrates all the parameters of the basis and is in a way the fingerprint of the basin.

Therefore the level or stage of the bankfull condition is crucial to an understanding of the river. Keep in mind that the bankfull stage is a geomorphic not a hydraulic feature.

The movement or migration of the channel across the valley floor results in the formation of a flood plain. The flood plain is defined as the flat area near the channel constructed by the river in the present climate and inundated at times of high flow.

This construction is carried out primarily by the growth or extension of point bars and so the top of the point bar is the elevation of the flood plain and is the elevation of the bankfull stage. Thus the elevation of the point bar is on of the best indicators of bankfull condition.

The next most important indicator of change from point bar to flood plain is a change in vegetation. A change in slope is an indicator, for example where a bank sloping upward becomes a vertical bank, or where a vertical bank grades upward to a sloping surface. Another indicator is a change in size or size-distribution of bank material, for example, a change from cobble to mixture of sand, cobbles and fine gravel.

In most regions of the country, there is an abandoned flood plain above the flood plain which is called a terrace.

Of all the parameters, bankfull width is the most easily adjusted and is sensitive to the input of water and sediment. The measurement of bankfull width should be done on a straight stream reach. At the same location, the average bankfull depth needs to be measured.

In several regions of the country, studies have document relationships between channel parameters and watershed area. Bankfull width, bankfull average depth and cross sectional area have been correlated to watershed area.

Relationships of bankfull width, average bankfull depth and cross sectional area to watershed area can be developed for reference streams. At 10 to 20 straight reaches located from the headwaters to larger order streams lower in the watershed, bankfull widths, average bankfull depths and cross sectional area are

measured in the field. The points of measurement are carefully located on topographic maps. The watershed boundaries above measure points can be drawn on the topographic maps and the watershed area estimated from the maps. Bankfull widths, average bankfull depths and cross sectional areas are plotted against watershed areas to develop the relationships.

These relationships can be used to evaluate the impacts of management on channel characteristics in study streams. A couple hypothetical examples will demonstrate the use of these relationships. First, a study reach extends from above to below a management activity that has removed the riparian vegetation and the stream banks are unstable. Bankfull widths, average bankfull depth and cross sectional are measured at straight reaches above, adjacent and below the management activity. These points are located on a topographic map and the watershed area above each study point is determined.

The watershed area for the point above the management area is entered into the reference relationships to get expected bankfull width, average bankfull depth and cross sectional area. The study stream width, depth and cross sectional area are compared with reference values. In this example, we will assume they are essentially the same.

However, the values for the adjacent and below reaches are significantly different from what would be expected for their respective watershed areas. The stream is wider and shallower than expected. This would suggest channel aggradation is occurring, heavy sediment deposition from the management activity or channel instability from the removal of riparian vegetation. Further investigation will be needed to determine the cause(s). The upstream reach was as expected suggesting that the impact is a reflection of the management activity.

Further investigation finds a low water bridge with a small culvert below the management area that is functioning as a sediment trap, the head of the sediment deposit is located near the top of the reach adjacent to the management area, and the removal of the riparian vegetation is facilitating the widening and shallowing of the stream. So, in this example, the problem rests with a poorly designed stream crossing and with the removal of riparian vegetation.

The second example addresses the concern about cumulative impacts of several management areas in a watershed. In this case, the relationships of bankfull width, average bankfull depth and cross sectional area are developed for the study stream. The relationships between the reference and study streams are compared to determine whether cumulative impacts have occurred or not.

In many streams, the comparison of particle size distributions within stream habitat units can reveal impacts. To do this requires the study reach have the same stream class as the reference reach. The particle size distribution for the reference stream is developed by taking a 100 count sample of streambed materials in a cross section of the stream and tallying them by particle size. The percent composition of each particle type (bedrock, boulders, cobble, gravel,

sand and silt/clay) is computed from the tally. The same process is used in the study reach (the study and reference reach need to be as similar as possible). Shifts in percentages in particle sizes can reveal impacts, for example a low percentage of sand in the reference site versus a high percentage of sand in the study site. (End of reference to Rosgen and Leopold 1991).

The detailed evaluation and comparison of bankfull widths, average bankfull depth, cross-sectional area, and particle size distributions would be used at the BMP Effectiveness III and IV levels of monitoring.

Also, BMP Effectiveness III and IV assessments could use detailed classification and mapping of channel types from headwaters to the mouth of the watershed. The channel type will vary with location on the stream. The reference stream will set the pattern of expected channel types as the stream descends to the mouth. If at a similar location along the study stream an unexpected channel class occurs in association with management, an impact of management might be indicated. For example, a D4 (braided) stream could be found where a C4 (single channel) stream would be expected. Or C5 (sand bottom) reach could be found where a C3 (cobble bottom) is expected.

At the BMP Effectiveness I and II levels of monitoring, comparing channels above, adjacent to, and below a management area may reveal changes in classes, which, through additional observations, can be linked to an activity. An observer knowledgeable about reference stream conditions may know what channel types should be present given the location in the study watershed.

Alaska Channel Classification

Channel types are used to distinguish the various parts of a stream system. They allow the user to define the characteristics of the channel and to accurately predict probable responses to natural and human influences, and thus are tools intended to complement a holistic approach to watershed management. However, channel types cannot be managed as isolated segments. A stream reach in one part of a watershed can be affected by activities taking place in a different part of the watershed, either upstream, downstream, or on adjacent land areas.

The Alaska Region Channel Type Classification is defined according to nine basic fluvial process groups (Paustian et al. 1992). These process groups describe the interrelationship among watershed runoff, landform, geology, and glacial or tidal influences on fluvial erosion and deposition processes. Individual channel type classification units within each process group are defined by physical attributes, such as channel gradient, channel pattern, stream bank incision and containment, and riparian plant community composition.

The Channel Type Classification System was developed with water-resource management needs in mind. Propagation of anadromous fisheries is the major beneficial use of water resources in Southeast Alaska (Paustian et al. 1992). Channel-type inventories provide key information on fish habitat utilization and capability and fisheries enhancement options in survey area watersheds.

Channel types also provide information on suitable stream-crossing locations and design criteria for road drainage structure. Channel types are used to evaluate potential sediment delivery and retention for cumulative watershed-effects analysis.

Classes are designated by a channel type name and the process group name.

Process Group Name. Channels bear the signatures of the processes that formed them. Channels formed and maintained by the same or similar fluvial processes are grouped for taxonomic purposes. Process groups reflect the long-term interaction of geology, landform, climate, and riparian vegetation. They characterize the basic interrelationships among the runoff, sediment transport, and vegetation patterns along stream banks.

Channel Type Name. Within each process group are a number of channel types that further define differences and describe individual channels. Channel types have fewer variables than the process group. This allows for more site-specific analysis and prescription for project plans. Some channel types may have one or more phases, or common variants, which influence management interpretations.

The following are the Process Group Names and associated Channel Types:

Estuarine Process Group
- Silt Substrate Estuarine Channel or Slough
- Narrow Small Substrate Estuarine Channel
- Narrow Large Substrate Estuarine Channel
- Large Estuarine Channel
- Broad Braided Glacial Outwash Estuarine Channel

Palustrine Process Group
- Narrow Placid Flow Channel
- Moderate Width Placid Flow Channel
- Shallow Ground Water Fed Slough
- Flood Plain Backwater Slough
- Beaver Dam/Pond Channel

Flood Plain Process Group
- Uplifted Beach Channel
- Foreland Uplift Estuarine Channel
- Narrow Low Gradient Flood Plain Channel
- Low Gradient Flood Plain Channel
- Wide Low Gradient Flood Plain Channel

Glacial Outwash Process Group
- Glacial Outwash Flood Plain Side Channel
- Large Meandering Glacial Outwash Channel
- Large Braided Glacial Outwash Channel
- Moderate Width Glacial Channel
- Cirque Channel

Alluvial Fan Process Group
- Moderate Gradient Alluvial Fan Channel
- High Gradient Alluvial Cone Channel
- Glacial Alluvial Cone Channel

Large Contained Process Group
- Low Gradient Contained Channel
- Moderate Gradient Contained Narrow Valley Channel

Moderate Gradient Mixed Control Process Group
- Narrow Mixed Control Channel
- Moderate Width Mixed Control Channel

Moderate Gradient Contained Process Group
- Narrow Shallow Contained Channel
- Moderate Width and Incision, Contained Channel
- Deeply Incised Contained Channel

High Gradient Contained Process Group
- Shallow Incised Muskeg Channel
- Shallow to Moderately Incised Footslope Channel
- Deeply Incised Upper Valley Channel
- Deeply Incised Muskeg Channel
- Shallow Incised Very High Gradient Channel
- Deeply Incised Mountainslope Channel
- Moderate/High Gradient Glacial Cascade Channel
- High Gradient Incised Glacial Torrent Channel

V* Pool Sediment

Lisle and Hilton (1992) developed a method to evaluate and monitor the supply of mobile sediment in gravel-bed streams and to detect and evaluate sediment inputs along a channel network by measuring fine sediment in pools. This method applies primarily to the BMP Effectiveness III and IV levels, and occasionally to Level II when coupled with other qualitative methods.

Mobile sediment tends to be concentrated in pools; thus the fraction of the pool filled indicates the supply of mobile sediment in natural, gravel-bed channels. During high flow, fine sediments in scour pools are remobilized. During waning flows, fine sediment is selectively transported from zones of relatively fast-moving water, such as riffles, and deposited in zones of slow-moving water, such as pools, where they cover a coarser substrate.

Fine sediments tend to move fast and can be flushed rapidly from streams. Therefore, their high concentration on a streambed can indicate widespread, chronic supplies or recent, local inputs (Platts and Megahan 1975). Abundant fines on the bed surface may indicate an increase in sediment supply (Dietrich et al. 1989). Finally, fine-sediment abundance can indicate a reduction in transport capacity without a compensating decrease in sediment supply.

Study reaches should have gentler channel gradients than adjacent reaches because gentle reaches tend to store more sediment and are sensitive to changes in sediment inputs. All study reaches should have well-developed riffle-pool sequences. The streambed surfaces should be covered by a layer of gravel, cobble and/or boulders. The procedure will not work well if the stream beds were originally sand, silt, or clay.

Fine sediment deposits and residual pool volumes are measured during low-flow periods. All depth measurements are made with a graduated 0.5-inch rod. The datum is the riffle crest below the pool. The water depth over the riffle crest is measured to define the datum. The residual pool is that pool volume remaining after a portion of the scour pool has been filled with fine sediment below the datum or riffle crest.

A series of transects is established across the pool and the depth to the streambed is measured at points along each transect. The residual pool depth is measured at the intercept of the rod with the water surface, minus the average water depth over the riffle. For example, if water depth is 2.5 feet and riffle water depth is 0.5 feet, the residual water depth is 2.0 feet.

At the same points, the rod is driven carefully into the fine-sediment deposit until it hits the buried substrate. The rod will stop or the observer will feel a vibration or resistance in the rod when it hits larger particles. The depth of fine sediment is the difference between depth to streambed and the depth to substrate. With a series of transects, the profile of the residual pool and fine sediment deposits can be plotted and the volume of each computed.

V* equals the volume of fine sediment divided by the sum of the residual pool volume and fine sediment volume, i.e., the fraction of the scour pool (the sum of the residual pool volume and fine sediment volume) filled with fine sediment. V* can be averaged for a series of pools in a reach below a management area and compared with the average V* for the reference reach above or in a nearby watershed. For example a V* for the reference reach of 0.1 and V* for the study reach of 0.3 would suggest the management activity supplied sediment to the stream.

Rosgen Channel Monitoring

Rosgen (personal communication, 1993) uses the following method to monitor channel and stream-bank erosion, which is appropriate for BMP Effectiveness III and IV monitoring. To isolate changes in channel stability and sediment supply caused by climate, geology, and morphology from those caused by management, monitoring should use one of the following three approaches:

1. Compare study reach with a reference reach (control).
2. Compare above versus below impact area for same stream type.
3. Compare before versus after management activity.

The field methods for evaluating channel stability and sediment supply are:

Channel Stability—Stability is assessed by measuring vertical and lateral changes in the stream channel.

> ***Vertical Stability*** (Aggradation/degradation)
> To determine if the stream is either down cutting (degrading), filling (aggrading), or is stable, permanently monumented cross-sections are used on at least one riffle and pool segment of a reach (or step vs pool segment if a step/pool stream type).

The bench mark for the cross-section should be located on a stable site above the active channel (Figure 18). An elevation cross-reference is often needed if one side of the cross-section is located on an unstable slope. An elevation bench mark is set, not necessarily tied into an absolute elevation, but rather a relative elevation set at 100 feet. To mark the ends of the transect, two holes are dug and filled with half a bag of premixed concrete. A 10-inch stove bolt is placed flush in the concrete before it sets. The transect markers need to be at the same elevation.

The profile of the stream cross-section is measured with a rod from a "leveled" tape line (Grant et al. 1992). The cross-section should be re-surveyed annually and/or following stormflow/snowmelt runoff events. Any "high water" marks need to be indicated on the cross-section. The cross-section profile is plotted for each measurement and compared to previous profiles.

Use Sag-tape Method as an alternative to using level/tripod survey instruments (Grant et al. 1993) as follows:

1. Locate permanent bench mark on both sides of stream.
2. Stretch tape tightly with spring clamp and level tape, i.e. tape at same elevation as reference bolt on bench mark.
3. Read distance and elevation reading of rod intercept with tape.
4. Measure major features such as:
   a. bench marks
   b. terraces/floodplains
   c. bankfull marks
   d. banks
   e. edges of water
   f. differences in bed configurations across bed
   g. thalweg
   h. inner-berm features

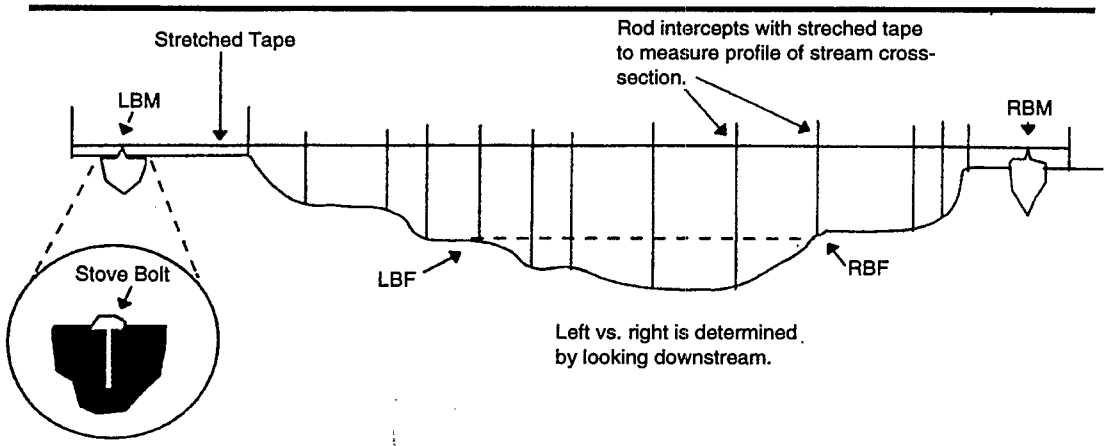

Figure 18. Example of a permanent cross-section with bench mark locations and points of measurement.

Lateral Stability

To determine the rate and magnitude of bank erosion, bank "pins" and/or bank profiles are installed on sites representative of the stream banks on the river (Figure 19). These should be stratified and installed on outside bends and straight reaches with different erodibility along stream segments of the same stream type. Erosion rates can be expressed in feet/year, cubic yards/year, or total tons/stream reach.

The erosion-pin method involves inserting two or three smooth rods 4 to 5 feet long and 0.3 to 0.5 inch in diameter, horizontally into the banks at a permanent cross-section (Figure 19). Periodically, or at least before and after storms, the distances between the end of the rods and the bank are measured. This not only reveals the rate of lateral migration, but also indicates if dimensions (such as bankfull width) and elevations are changing.

The bank profile procedure involves (Figure 20):

(1) Installing a permanent cross-section over the bank profile site.
(2) Installing a permanent toe pin (rod) offset and directly adjacent to the study bank.
(3) Placing rod level on survey rod that is set on toe pin.
(4) Stabilizing rod with either a tripod or a frame to attach to bank to hold rod "plumb."
(5) Measuring horizontally with tape rule from vertical rod to bank. (Measure at frequent intervals to describe bank dimensions and features.)
(6) Plotting data to display profile for each survey.
(7) Comparing with previous surveys annually or following storms (Figures 21 and 22).
(8) Computing mean erosion rate.

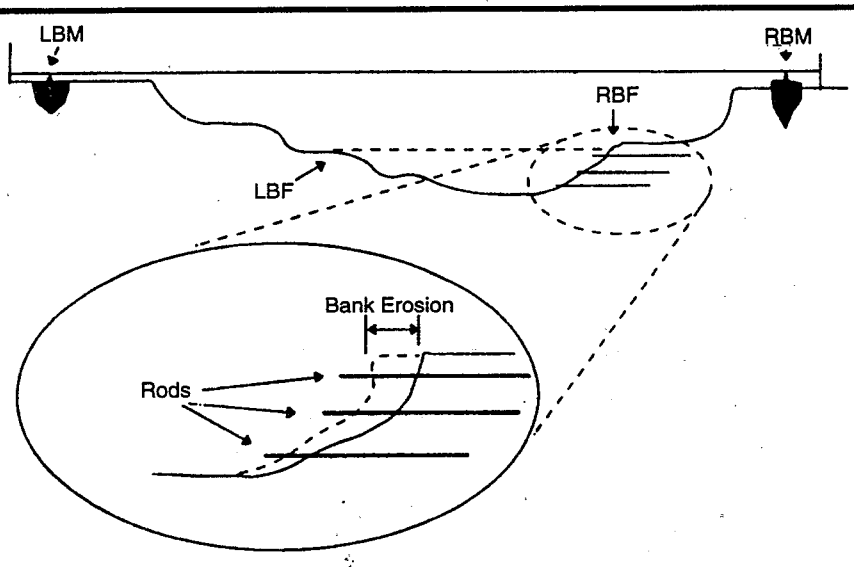

Figure 19. Measuring bank erosion with bank pins.

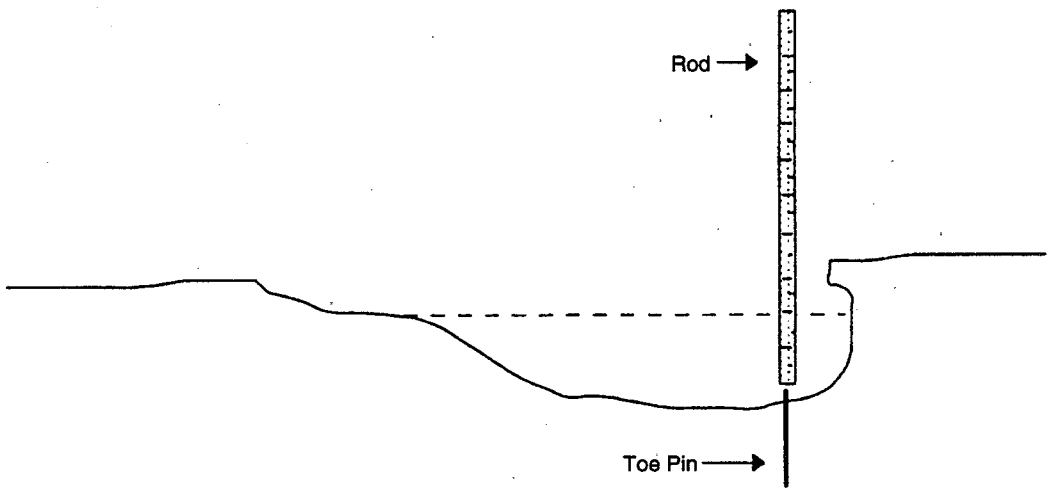

Figure 20. Bank profile procedure using toe pin and survey rod.

Bed Material Size Distribution

To observe a shift in bed-material size distribution use one of the following:

(1) Transect to obtain existing and departures from particle size distributions by measuring 100 particles from bed material at riffles, pools, and special features of interests (spawning reds, fish-habitat structures, etc.).

(2) Riffle Armor Index (Kappesser 1993) where fine material deposition changes can be quantified.

(3) Freeze core sampling and other measures documented in MacDonald et al. (1991).

If a pebble-count method is used without a permanently established transect for resurvey, then samples should be taken in proportion to the frequency of riffles and pools. For example, if 80 percent of the channel is composed of riffles, then 80 percent of the cross-sections sampled should be in riffles. Permanent transects are preferred for a higher level of detail, to make replication easier, and to reduce sample error.

Pebble Counts

Pebble counts can be used at all levels of monitoring to determine if the composition of stream-bed surfaces differs between reference and study streams, which can reflect differences in sediment supply. The pebble count determines the proportion of bed material less than a given size. It can be done in wadable streams with beds of coarse material. A pebble count consists of selecting at least 100 particles located by the toe of the observer's boot as he/she walks along a transect from bankfull to bankfull stage (Leopold 1970). A particle has three axes that are mutually perpendicular: long, short, and intermediate in length. The intermediate axis of each particle is measured. Pebbles down to 2 mm can be easily measured, but smaller sizes are judged by feel, using soil-classification methods.

Particle sizes are tallied by the following classes: <2 mm, 2-4 mm, 4-8 mm, 8-16 mm, 16-32 mm, 32-64 mm, 64-128mm, 128-256 mm, 256-512 mm, 512-1024 mm, 1024-2048 mm and > 2048 mm. A cumulative size distribution curve is created by plotting the cumulative percent fines over particle size class.

Reference and study reaches should be compared where reaches are of the same stream size and expected geomorphic classification. Two hypothetical cumulative size-distribution curves are presented in Figure 23. The study stream contains many more fine particles than the reference stream. If on-slope monitoring of the management activity traces sediment to the stream and no other source can be identified, then the differences can be attributed to the management activity.

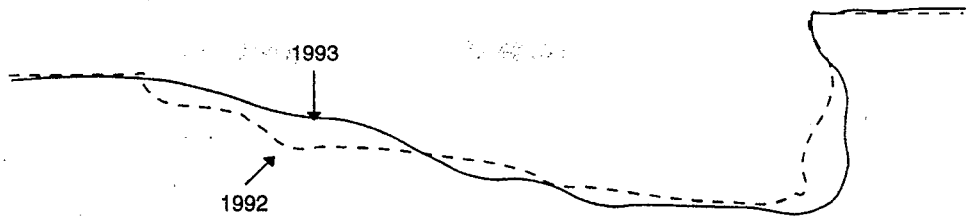

Figure 21. Comparison of relative elevations for cross-sections for a 2-year period.

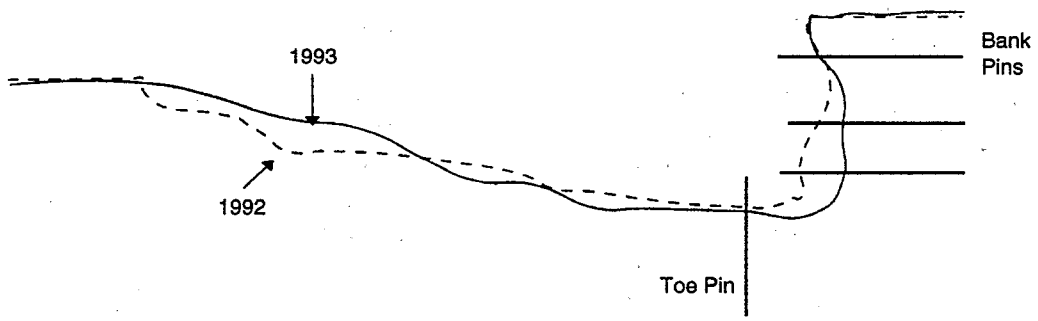

Figure 22. Combined cross-section and bank erosion study for a 2-year period.

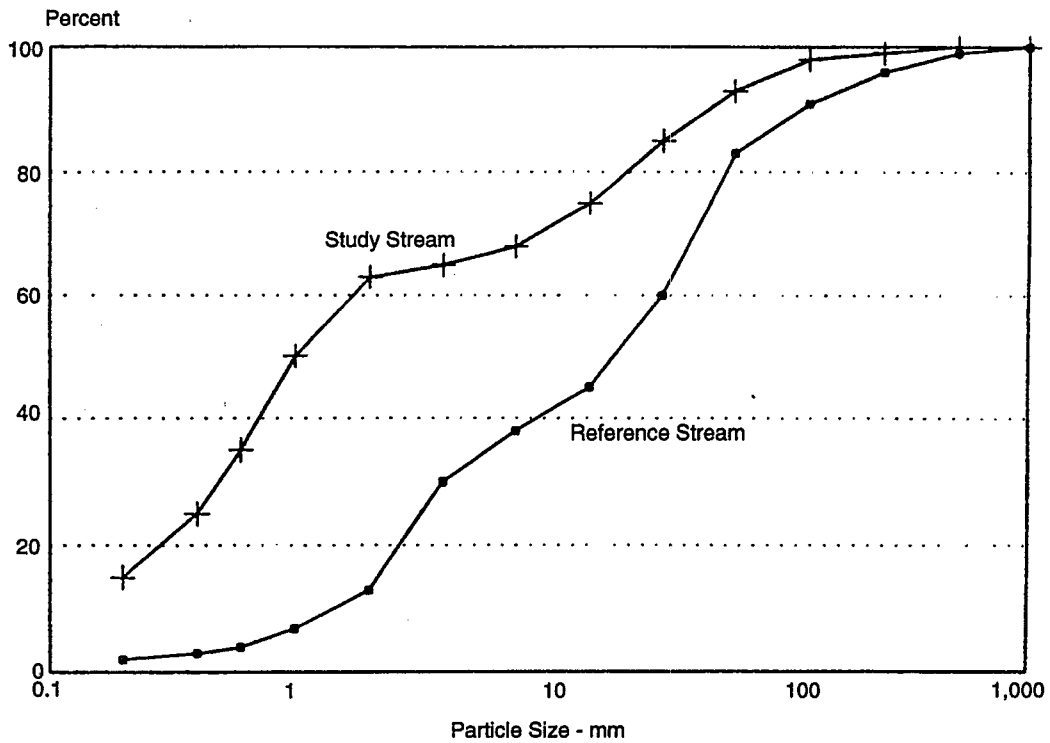

Figure 23. Comparison of two hypothetical cumulative sediment size curves between a reference and study stream.

Habitat Monitoring Methods

This section summarizes various habitat monitoring and classification methods to assist the readers in selecting the method most appropriate to their needs. In addition to delineating habitat types for comparison in habitat monitoring, habitat classification defines habitat niches to sample for benthic macroinvertebrates, fish, and aquatic vegetation. Efforts to evaluate habitat should describe current habitat and the extent to which it resulted from human activity as opposed to natural processes at the site.

For more detailed information, the reader is referred to the original literature. As with the biological monitoring methods, some of the discussions are excerpts from the source papers and others are summary statements. No distinction is made, but all are clearly documented.

Habitat Monitoring Methods by Monitoring Level

The habitat monitoring methods abstracted here evaluate aquatic and terrestrial habitat conditions associated with benthic macroinvertebrates and fish. Specific monitoring methods are recommended for each level of monitoring:

	Page
BMP Effectiveness I	
Empirical/perceptual	116
Rapid Bioassessment Protocol	116
Streamwalk	118
T-Walk	119
BMP Effectiveness II	
Rapid Bioassessment Protocol	116
Basinwide Estimation of Habitat	123
Hankin-Reeves	123
BMP Effectiveness III	
Rapid Bioassessment Protocol	116
Bisson	120
Hankin-Reeves	123
Forest Service R-5, R-1, R-4 FHR	124
Platts Riparian Habitat	126
Ohio-QHEI	126
EMAP	118
BMP Effectiveness IV	
Rapid Bioassessment Protocol	116
Hankin-Reeves	123
Bisson	120
Forest Service R-5, R-1, R-4 FHR	124
Platts Riparian Habitat	126
Ohio-QHEI	126
EMAP	118

Empirical/Perceptual

At the BMP Effectiveness I level, a trained professional needs to know the key habitat needs of invertebrates and fish, the habitat types, the prevalence of each habitat type, and the quality of the reference streams. The observer also needs to know the kinds and distribution of habitat types; the amount, type, and overhang of stream bank vegetation; bank angle and stability; pool depths; amount and distribution of large woody debris; the amount of stream surface shading; type and density of riparian vegetation; substrate type; percent embeddedness or percent fines in the substrate; etc. To supplement the visual picture, the observer may take notes or complete a form on reference habitat conditions and may even take photographs of the reference habitat conditions to assist in the comparing the study stream with the reference stream.

The key habitat elements will vary among ecoregions and the types of fish and benthic macroinvertebrate communities. In some ecoregions and for some fisheries, key habitat requirements have been developed that can be used in assessing the habitat quality of reference and study streams. As State water-quality agencies develop narrative and numeric criteria, key habitat criteria will be identified and should be used.

The observer mentally compares the study stream habitat with reference stream habitat, judging whether they are similar or whether the study stream habitat is impaired. Then the observer tries to determine whether the impairment was caused by the management situation under evaluation.

Rapid Bioassessment Protocol—Habitat

The Rapid Bioassessment Protocol—Habitat (RBP) is being adapted and adopted by State water-quality agencies. RBP Habitat applies to each RBP biological monitoring level because it is flexible. The RBP Habitat Assessment will be adapted to the ecoregions, invertebrates and fisheries of each State. If the State has developed RBP habitat elements, forest water-quality studies should use them in evaluating habitat responses to management.

According to Plafkin et al. (1989), the habitat assessment matrix (Table 8) is based on the "Stream Classification Guidelines for Wisconsin" developed by Ball (1982) and "Methods of Evaluating Stream, Riparian, and Biotic Conditions" developed by Platts et al. (1983). Because this habitat assessment is intended to support biosurvey analysis, the various habitat variables are weighted to emphasize those most biologically significant. All variables are evaluated for each station studied and rated according to a numerical system. The ratings are totalled and compared to a reference for final habitat ranking. Scores increase as habitat quality increases. To ensure consistency in the evaluation procedure, descriptions of the physical variables and related criteria are included in the rating form.

Reference conditions are used to normalize the assessment to the "best attainable" situation. This is critical because stream characteristics vary dramatically among ecoregions. Other habitat assessment systems may be used, or a more precise quantitative approach may be taken. However, the importance of a holistic habitat assessment to enhance the interpretation of biological data cannot be overemphasized.

Habitat variables pertinent to the assessment of habitat quality are separated into three principal categories: primary, secondary, and tertiary. Primary variables are those that characterize the stream "microscale" habitat and exert the greatest direct influence on the structure of the indigenous communities. These variables, which comprise characterization of the bottom substrate and available cover, and estimates of embeddedness, and flow or velocity and depth regime, have the widest score range (0-20) reflecting their contribution to habitat quality. The secondary variables define the "macroscale" habitat, i.e., channel morphology characteristics such as channel alteration, bottom scouring and deposition, and stream sinuosity. These variables have a score range of 0-15. Tertiary variables define riparian and bank structure and include: bank stability, bank vegetation, and streamside cover, variables most often ignored in biosurveys. These variables have a score range of 1-10.

Instream habitat is evaluated first, followed by channel morphology, and finally structural features of the bank and riparian vegetation. Stream segment length or area assessed will vary with each site. Generally, primary variables are evaluated within the first riffle/pool sequence or the immediate sampling area, such as in fish sampling.

Secondary and tertiary variables are evaluated over a larger stream area, primarily upstream, where conditions will have the greatest impact on the community being studied. The actual habitat assessment process involves rating the nine variables as excellent, good, fair or poor, based upon criteria included on the Habitat Assessment Field Data Sheet.

Table 8. RBP Habitat Rating

Habitat Variables	Condition			
	Excellent	Good	Fair	Poor
Primary-Substrate & Instream Cover				
1. Bottom substrate and available cover	16-20	11-15	6-10	0-5
2. Embeddedness	16-20	11-15	6-10	0-5
3. Flow/velocity	16-20	11-15	6-10	0-5
Secondary-Channel Morphology				
4. Channel alteration	12-15	8-11	4-7	0-3
5. Bottom scouring and deposition	12-15	8-11	4-7	0-3
6. Pool/riffle, run/bend ratio	12-15	8-11	4-7	0-3
Tertiary-Riparian and Band Structure				
7. Bank stability	9-10	6-8	3-5	0-2
8. Bank vegetation	9-10	6-8	3-5	0-2
9. Streamside cover	9-10	6-8	3-5	0-2

A total score is obtained for each study stream and compared to a reference stream. The score for the study stream is expressed as a percent of that for the reference stream and the study stream is then classified on the basis of its apparent potential to support an acceptable level of biological conditions (Table 9).

Table 9. Comparability Assessment

Assessment Category	Percent of Reference
Comparable to Reference	> 90%
Supporting	75-88%
Partially Supporting	60-73%
Non-Supporting	< 59%

Environmental Monitoring and Assessment Program (EMAP)

The USEPA (1993) is in the process of developing and testing a more quantitative system for evaluating habitat and the chemical, physical, and biological condition of streams. Occasionally, this system may be used in BMP Effectiveness III and IV levels. EMAP is mentioned to make the reader aware that a more detailed habitat assessment system is available than RBP.

Streamwalk

Streamwalk is a BMP Effectiveness I method that provides a qualitative assessment of habitat conditions (Idaho WRRI 1992). The variables may vary, but in Idaho the following are evaluated:

STREAM CHARACTERISTICS (Measured or estimated)
- Depth (feet)
- Width (feet)
- Gradient from map (Feet/100 feet)
- Pools and riffles (percent of stream)

STREAM BANK (Estimated percent of all banks in each category)
- % of banks mostly covered with vegetation
- % of banks mostly not covered with vegetation
- % of banks stable (not eroding, slumping, or sloughing)
- % of banks unstable (eroding, slumping or sloughing)

STREAM BOTTOM (Estimate percent in each category)
- % Silt/Clay/Mud
- % Sand (up to .1")
- % Gravel (.1 to 2")
- % Cobble (2 to 10")
- % Boulders (over 10")
- % Bedrock (solid)

STREAMSIDE RIPARIAN VEGETATION WIDTH (Feet)
(Estimate to nearest 10 feet)
Left (looking downstream)___Right___

STREAMSIDE VEGETATION TYPE (check appropriate boxes)

	None/Sparse	Occasional	Common
Conifers	[ ]	[ ]	[ ]
Deciduous	[ ]	[ ]	[ ]
Shrubs	[ ]	[ ]	[ ]
Grasses	[ ]	[ ]	[ ]

Vegetation Appears: Natural [ ] or Cultivated [ ]

Extent of vegetative canopy shading the water surface
[ ] 0-25% [ ] 25-50% [ ] 50-75% [ ] 75-100%

Extent of artificial bank protection
[ ] 0-25% [ ] 25-50% [ ] 50-75% [ ] 75-100%

Presence of logs or large woody debris in stream
[ ] None [ ] Occasional [ ] Common

Presence of other organic debris (small limbs, etc.) in stream
[ ] None [ ] Occasional [ ] Common

By applying the procedure to both study and reference streams, quality of habitat can be compared. Based upon experience, the observer can judge whether the study and reference streams are similar or whether habitat is impaired in the study stream compared to the reference stream.

Ocular estimation of some variables may be difficult and results have varied among observers and sampling periods. This can influence reliability of the data and repeatability of the methods.

T-Walk

Thalweg Walk (T-Walk) uses qualitative data to evaluate the effectiveness of management in protecting habitat, channel stability, and macroinvertebrates (Ohlander 1991). The method deals chiefly with stream habitat condition and channel stability, coupled with qualitative sampling of invertebrates and an estimate of invertebrates as fish food, based upon substrate size distribution.

The method primarily involves monitoring above and below the management site and occasionally includes reference streams. The recommended length of stream segments to evaluate is 600 feet. Substrate size, sand infill, vegetation, maximum and minimum water depth, and flow velocity are measured at 30 to 100 points along the segment.

Based upon the substrate composition, a Tarzwell substrate ratio is determined. The Tarzwell substrate ratio is a dimensionless measure of fish-food organisms produced on different substrates, indexed to sand as the least productive. Tarzwell (1937) quantified fish food production for 25 different substrate conditions.

Channel morphology is characterized by evaluating stream slope, sinuosity, bankfull width and depth, channel depositional features (types of bars), bank conditions, and stability.

Riffle insects are evaluated by sampling 10 cobbles, and identifying and enumerating the various invertebrates.

Adjacent land uses and conditions are assessed.

Compared with reference conditions, the study stream characteristics are rated as: robust, adequate, diminished, impaired, precarious, or catastrophic. These qualitative ratings are used to judge whether management has been effective. The system is designed to identify early those management activities that are clearly not effective.

Bisson Habitat

Bisson et al. (1981) presented a system for classifying habitat in small streams that can be used in BMP Effectiveness III and IV assessments. Fish habitat is classified according to location within the channel, pattern of water flow, and nature of flow controlling structures. Riffles are divided into three habitat types: low gradient riffles, rapids, and cascades. Pools are divided into six types: *secondary channel pools, backwater pools, trench pools, plunge pools, lateral scour pools, and dammed pools. Glides, the last habitat type, are intermediate in many characteristics between riffles and pools.*

Bisson et al. (1981) habitat classification system is as follows:

> ***Low gradient riffles** are shallow (< 20 cm deep) stream reaches with moderate velocity (20-50 cm/sec) and moderate turbulence. Substrate is usually composed of gravel, pebble, and cobble-sized particles (2-256 mm). An upper gradient limit for this habitat type was arbitrarily set at 4 percent.*
>
> ***Rapids** possess a gradient greater than 4 percent with swiftly flowing water (> 50 cm/sec) having considerable turbulence. The substrate of rapids is generally coarser than the substrate for low gradient riffles, and during low streamflow conditions large boulders typically protrude through the surface.*
>
> ***Cascades** are the steepest. Unlike rapids, which have an even gradient, cascades consist of a series of small steps of alternating small waterfalls and shallow pools. The usual substrate is bedrock or an accumulation of large boulders; however, this habitat type is occasionally found on the downstream side of woody debris.*

During low streamflow conditions there are six pool types, which are associated with the presence of bedrock outcroppings, large rocks, or large tree stems and rootwads in the channel.

***Secondary channel pools** are those that remained within the baneful margins of the stream after freshets. During the summer, most of these pools will usually disappear, and those remaining have little flow through them. Secondary pools are usually associated with gravel bars, but many contain sand and silt substrates.*

***Backwater pools** are found along channel margins and are caused by eddies behind large obstructions such as rootwads and boulders. This pool type is often quite shallow (< 30 cm) and tend to be dominated by fine grained substrates. Like secondary channel pools, backwater pools possess very low current velocities.*

***Trench pools** are long, generally deep slots in a stable substrate. Channel cross sections are typically U-shaped with coarse-grained bottom flanked by bedrock walls. Current velocities in trench pools are the swiftest of any pool type and the direction of flow is the most uniform.*

***Plunge pools** occur where the stream passes over a complete or nearly complete channel obstruction and drops vertically into the streambed below, scouring out a depression. This type of pool is often large, quite deep (> 1 m), and possesses a complex flow pattern radiating from the point of water entry.*

***Lateral scour pools** differ from plunge pools in that the flow is directed to one side of the stream by a partial channel obstruction. Often an undercut bank is associated with this type of pool.*

***Dammed pools** consist of water impounded upstream from a complete or nearly complete channel blockage. Typical causes of dammed pools are debris jams, rock landslides, or beaver dams. Depending on the size of the blockage, dammed pools could be very large. Water velocity in dammed pools is characteristically slow and substrates tend toward smaller gravels and sand.*

***Glides** are a third general habitat category which possesses attributes of both riffles and pools. Glides are characterized by moderately shallow water (10-30 cm) with an even flow that lacks pronounced turbulence. They are usually located at the transition between a pool and the head of a riffle. Glides are occasionally found in long, low gradient stream reaches with stable banks and no major flow obstructions. The typical substrates are gravel and cobbles.*

***Cover** includes eight distinct kinds of cover for fish. These include three kinds of wood debris—rootwads, large debris (tree stems), and small debris (branches, twigs,etc.)—that differ in the amount of overhead cover and flow*

modification they provide within the channel. Overhanging terrestrial vegetation and undercut banks are two kinds of cover that are governed by the condition of the riparian zone. Water turbulence acts as cover when the presence of bubbles prevent a clear view of water beneath. Rocks function as cover in two ways: (1) by providing overhanging ledges and (2) by providing crevices for hiding. Finally, deep water is itself a form of cover from non-diving terrestrial predators. The primary function of cover during the summer is assumed to be protection from predation.

Each stream reach is surveyed on foot and the location of different habitat types, as well as significant flow controlling structures, is drawn to scale on a map. Contour lines based on depth measurements are drawn within pools to enable volume estimation. Wetted surface areas are determined by counting squares on gridded paper that is superimposed on the maps. Axial length is figured as the distance along the thalweg or greatest linear dimension of a habitat unit parallel to the direction of flow. Reach summaries are constructed by summing the lengths, areas, and volumes of each habitat type and expressing each group as a percentage of the total.

The amount of cover in each habitat is rated on a relative abundance scale of 0-3, where a score of zero indicates that the particular kind of cover is essentially absent and a score of three indicates a very abundant condition. Substrate is noted as according to the predominant type.

Fish populations are sampled by isolating individual habitat types with block nets and electrofishing the habitat three times. The fish population is characterized as to species, biomass, and age.

The utilization of habitat types by species and age classes is evaluated. In order to quantify habitat utilization by species and individual age classes, it is necessary to relate the fraction of the pollution found within a particular habitat type to the relative abundance of that habitat type in the stream. The formula used is based upon the electivity index of Ivlev (1961):

$$\text{Utilization} = \frac{\text{habitat specific density} - \text{average total density}}{\text{average total density}}$$

Where:

habitat specific density = average density in the habitat type in question

average total density = average density over the entire stream reach, all habitats combined

Values for habitat utilization coefficients can theoretically range from minus one, indicating total non-use of a habitat type, to positive infinity as greater proportion of the population resides in the habitat type on interest. A value of zero indicates that the population occurs in the habitat type in proportion to that type's abundance in the stream.

Hankin-Reeves Habitat

Hankin-Reeves visual estimation techniques allow surveyors to inventory a preselected set of habitat characteristics in an entire watershed. Habitat sampling is done two steps (Hankin-Reeves 1988): (1) individual habitat units are classified and visually estimated by type and habitat characteristics (e.g., surface area, substrate composition), (2) visual estimates are paired with precise measurements taken at a predetermined number of units (at least 10 for each habitat type) to develop calibration ratios. Sampling teams typically consist of two people: an estimator and a recorder.

Before heading for the field, sampling teams first: (1) select the classification system they will use to identify habitat types (e.g., Bisson et al. 1981); (2) determine the habitat characteristics they will survey; and (3) stratify the study area into survey units (reaches) based on gradient, confluence of same-order channels, or other distinctive features. The team then decides how many of the habitat units in each habitat type will be measured. The number of units to be measured is based on the consistency between estimates and measurements of habitat areas and on the expected number of units of each habitat type. For example, a smaller number of pools would need to be measured if the team always overestimated pool areas than if it sometimes overestimated and other times underestimated. And if the habitat type is rare, the team may need to measure most or even all the units of that particular type to ensure that the number of paired measurements and estimates is at least the recommended minimum of 10.

The area of measured habitat units is calculated as the product of mean width and length. Mean width is derived from measurements taken at three or more locations parallel to the thalweg along the length of the unit. The interval between width measurements will depend on the complexity of the unit. A 5-meter interval may be appropriate for a 15-meter stretch of straight riffle. However, that same length of stream in an irregularly shaped pool might require a 2-meter measurement interval. Estimates and variances of total habitat area for each habitat type within a stream reach are calculated using equations developed by Hankin and Reeves (1988).

The habitat distribution can be compared between study and reference streams. The Hankin-Reeves method would be used to determine the habitat types and areas, the relative mix, and habitat distribution along the stream. If key habitat types are lower in quantity and quality in the study stream than in the reference stream, habitat may have been impacted by management in the study watershed. Indications of habitat changes might include differences in the riffle/pool ratio, number and size of pools, etc. Evaluating such shifts in habitat can reveal the types of management impacts.

Basinwide Estimation of Habitat

Dolloff et al. (1993) modified the Hankin-Reeves method to achieve more consistent interpretations of habitat types. This method is best suited for BMP Effectiveness II. They limited habitat types to four: pool, riffle, cascade, and complex. Using the Hankin and Reeves (1988) more detailed classifications leaves it to

personal judgment to determine where habitat types change. In this method, if a pool and riffle exist side by side along the stream, the dominant habitat type is assigned to the stream segment.

The survey reaches are defined by gradient, confluence of same-order channels, or other distinctive features. The habitat units are estimated for area and a subset of habitat units is measured to compute area.

Additional habitat information is collected, including average and maximum depth, dominant substrate, degree of embeddedness, number of pieces of large woody debris by size classes, and features that are likely to influence fish populations, such as landslides, tributary junctions, bridges, trail crossings, and significant changes in riparian vegetation. The total area of each habitat type is estimated using equations from Hankin and Reeves (1988) and Dolloff et al. (1993).

USDA Forest Service Region 5/Region 1/Region 4 FHR Stream Habitat Classification and Inventory Procedures

This method is designed to survey both fish habitat and fish populations (Overton et al. 1990). It is applicable to BMP Effectiveness III and IV assessments. Population is estimated as described in Hankin and Reeves (see page 136). The method presents a classification system that has 22 habitat types. Many are the same as the Bisson et al. (1981) habitat types, including:

1. *Low gradient riffles*
2. *High gradient riffles (rapids)*
3. *Cascade*
4. *Secondary channel pool*
5. *Trench/chute*
6. *Plunge pool*
7. *Dammed pool*
8. *Glide*

Several of the Overton et al. (1990) habitat types are refinements of the Bisson types, reflecting the channel feature associated with the type.

9. *Backwater pool—boulder formed*
10. *Backwater pool—root wad formed*
11. *Backwater pool—log formed*
12. *Lateral scour pool—log formed*
13. *Lateral scour pool—root wad formed*
14. *Lateral scour pool—bedrock formed*
15. *Lateral scour pool—boulder formed*

Overton et.al. (1990) identified these additional habitat types:

16. *Run*—Swiftly flowing reaches with little surface agitation and no major flow obstructions. Often appears as flooded riffles. Typical substrates are gravel, cobble, and boulders.

17. *Step run*—A sequence of runs separated by short riffle steps. Substrates are usually cobble and boulders.

18. *Mid-Channel pool*—Large pools formed by mid-channel scour. The scour hole encompasses more than 60% of the wetted channel. Water velocity is slow and the substrate is highly variable.

19. *Edgewater*—Quiet, shallow area found along the margins of the stream, typically associated with riffles. Water velocity is low and sometimes lacking. Substrate varies from cobbles to boulders.

20. *Channel confluence pool*—Large pools formed at the confluence of two or more channels. Scour can be due to plunges, lateral obstructions or downscour at the channel intersections. Velocity and turbulence are usually greater than those in other pool types.

21. *Pocket water*—A section of swift stream containing numerous boulders and other large obstructions that create eddies or scour holes (pockets) behind obstructions.

22. *Corner pool*—Lateral scour pools formed at a bend in the channel. These pools are common in lowland valley bottoms where stream banks consist of alluvium and lack hard obstructions.

This method uses both Bisson habitat typing and Hankin and Reeves estimation/ measurement correction factors. The primary difference between this method and Hankin and Reeves is the spatial scale of interest. USFS R5/R1/R4 FHR was designed primarily to distinguish microhabitat features and associated fish usage. For instance, Overton found that salmonid fish species distribution was distinctly different between the various pool types surveyed. He developed the method originally for anodromous salmonid streams in northern California coastal watersheds but has expanded its usage to high Sierra Nevada, Rocky Mountain, and intermountain western watersheds containing both resident and anadromous salmonids.

In assessing habitat for a stream reach or an entire basin, the intent is to gather information that will adequately describe the area of interest. Conducting a habitat inventory can be time-consuming, so work must be carried out quickly and efficiently. The level or scale of inventory to be used depends on the project objectives. The system is used at two scales: streams in a watershed or a stream segment associated with a project. Basin-level habitat classification is on the scale of a stream's naturally occurring pool-riffle-run units, where habitat unit size depends on stream size and order. Generally in a basin-level inventory, homogeneous areas of habitat that are as long as or longer than one channel width are recognized as distinct habitat units. On the other hand, project-level habitat assessment uses a scale of less than one channel width for use on reaches of intense management or study. Project-level habitat typing is used to quantify changes in habitat as the result of fish-habitat-enhancement projects or to evaluate management impacts on habitat.

Habitat typing can be done efficiently by two or three field people. However, describing and measuring all 22 habitat types is labor intensive; an average of one mile per day can be completed by trained surveyors. Decisions are best reached by a consensus among the team members after a discussion of the facts, thereby balancing out the biases inherent in each observer and ensuring quality data.

The basic method of habitat typing is relatively simple. Start at the mouth of a stream and work upstream to establish a known starting point. Measure mean length and width of each unit using a tape, rod, optical rangefinder, or hip chain. Three to five width measurements are sufficient. Along each width measurement transect, use a graduated leveling rod (or similar device) to take several depth measurements from bank to bank and estimate mean depth. If a significant portion (>10%) of the measured habitat includes exposed boulders and/or islands, that portion should be estimated and subtracted from calculations of area (total area - exposed area = wetted area). Data on other variables, such as stream substrate, in-stream cover elements and abundance, canopy cover, riparian quality, etc., can be collected along with the habitat type data.

The habitat data can be used to compare reference and study streams as previously described in the Hankin/Reeves and Dolloff/Hankin/Reeves discussions. Again, for this to be used as a monitoring tool, it is important to link habitat changes with the specific effects of particular management activities.

Ohio-QHEI

The Ohio Environmental Protection Agency has developed a qualitative habitat evaluation index (QHEI) to evaluate habitat quality for fish and macroinvertebrates (Rankin 1989). Ohio-QHEI is an example of habitat monitoring methods being developed by State water-quality agencies. It is being used at the BMP Effectiveness III level but is also applicable to level IV.

Habitat measurements taken include: substrate type and quality, instream cover type and amount, channel quality (sinuosity, development, channelization, and stability), riparian/erosion (width, floodplain quality, and bank erosion), pool/riffle (maximum depth, current available, pool morphology, riffle/run depth, riffle substrate stability, and riffle embeddedness), and gradient. Each measurement is scored and the sum of the scores is the habitat quality index. The QHEI has been correlated with biological indexes.

Platts Riparian Habitat

Riparian habitat is closely linked to stream-channel habitat and directly influences the biotic health of streams. Platts et al. (1987) developed a monitoring methodology for evaluating riparian habitat that is applicable to BMP Effectiveness III and IV assessments. Vegetation in the riparian ecosystem, including vegetation on the stream bank and flood plain, is evaluated because it helps stabilize stream banks, control nutrient cycling, reduce water velocity, provide cover and food for aquatic biota, and shade the stream.

Riparian vegetation is characterized by type, density, use by animals, overhanging vegetation, streamside stability, and cover. Riparian soils are classified according to flood plain geomorphology, valley-bottom conformations, soil morphology, stream bars, channel levees, adjacent wetlands, and riparian meadows.

Vegetative canopy closure and density are evaluated using a concave spherical densiometer. Light intensity and stream-surface shading are estimated to gauge the inputs of solar energy inputs into the stream.

Stream bank and channel aggradation, degradation, and morphology are measured using channel cross-sections. Banks, active channel, bank undercuts, stream bottom, and bank heights are some of the features evaluated.

Organic debris is mapped and measured. Large woody debris is classified by size:

1. Fines—needles, twigs, and pieces < 5 cm average diameter
2. Coarse—branches, limbs and pieces 5-20 cm in diameter and up to 2.5 m in length
3. Heavy—logs, trees, branches, stumps and pieces > 20 cm in diameter
4. Debris jams

Large woody debris (heavy) is further classified as complete bridge, collapsed bridge, ramp, and drift.

Platts et al. (1987) discuss in detail monitoring and evaluating methods, statistical design, and analysis of data.

Biomonitoring Methods

Biological monitoring methods are concerned with fish and benthic macroinvertebrates. Aquatic life is continually exposed to influences on streams and its abundance and diversity reflect these impacts.

This section acquaints the reader with several of the biomonitoring methods available as an aid in selecting the method appropriate to his or her monitoring effort. For more detailed information, the reader is referred to the original literature.

If the State agency has established biological monitoring methods, these should be used to ensure comparable data. Many State water-quality agencies use or modify Rapid Bioassessment Protocols (RBP) (Plafkin et al. 1989), while others develop their own methods.

Many fish and invertebrate metrics are being used to evaluate water-quality impacts (EPA 1993). As a help in preparing this document, two workshops were held at which experts in fish and invertebrate metrics offered their best professional judgment of metrics to consider for small forest streams. The following metrics for invertebrates were identified by both EPA (1993) and the workshops:

EPT/Chironomid + EPT
EPT Index (number of EPT taxa)
Percent EPT taxa
Total Taxa Richness
Chironomid Richness
Hilsenhoff Biotic Index
Percent dominant taxa
Index of Community Integrity
Community Loss Index
Jaccard Coefficient of Community
Percent Shredders
Percent Filters
Percent Scrapers
Ratio Scrapers/Filter Collectors
Total Abundance

The reader is referred to EPA (1993) for information on these metrics. The workshops also identified additional metrics judged as having potential for forest streams:

Biotic Condition Index (Winget and Mangum 1979)
Biomass
Voltinism (Number of generations annually for a species)
Percent Tolerant taxa
Percent Intolerant taxa

For invertebrate monitoring, biological expertise should be consulted to identify metrics to consider for a monitoring project. The key to invertebrate monitoring and analyses is sampling procedures and identification of taxa. The metrics are results of data analysis. The response of metrics to forestry activities should be compared to reference conditions to evaluate effectiveness.

EPA (1993) presents a long list of fish metrics for large western rivers, depauperated fish assemblages, and others to consider. Many of these fish and invertebrate metrics are different than those used in the Rapid Bioassessment Protocol discussed in this chapter. These metrics demonstrate the need to adapt the Rapid Bioassessment Protocol or other biomonitoring methods to ecoregions.

Biological Biomonitoring Methods by Monitoring Level

Monitoring methods are listed below, followed by abstracts of the various methods recommended for each monitoring level. These methods represent a cross-section of biological monitoring techniques being used.

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BMP Effectiveness II	
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Hankin-Reeves (Fish)	136
North Carolina Standardized Ten Sample Method	145

Empirical/Perceptual

At the BMP Effectiveness I level, a trained observer needs to know the benthic invertebrate communities found in the reference streams. The observer also needs to know which families of invertebrates are intolerant or tolerant to NPS impacts from forestry as well as the functional feeding groups (shredders, scrapers, etc.) and their habitat niches.

Empirical/perceptual monitoring of invertebrates relies on visual observations. The observer picks up rocks, leaves, and woody debris from key habitat in the stream bed and determines which families of invertebrates are present. Comparing these observations with knowledge of life forms in the reference streams, the observer judges whether the study stream has been obviously impaired or not. Indicators of impairment will be the absence of intolerant invertebrates, the dominance of tolerant species, the relative abundance of invertebrates, and the presence or absence of functional feeding groups.

Rapid Bioassessment Protocol I—Benthic Macroinvertebrates (RBP I)

RBP I is a screening or reconnaissance assessment of benthic macroinvertebrates that involves systematically documenting specific visual observations made in the field by trained professionals (Plafkin et al. 1989). This system is appropriate for the BMP Effectiveness I level. RBP I is used to identify obviously impacted and non-impacted streams and those requiring further investigation—in other words, the grey area between the obviously impaired and the obviously non-impaired conditions. To assess whether impairment exists once a grey area has been identified, RBP II or RBP III (discussed below) can be used to assess the integrity of the benthic invertebrate community.

The biosurvey component of RBP I includes qualitative sampling of benthic macroinvertebrates, supplemented by a preliminary field examination of other biota (periphyton, macrophytes, slimes, and fish) (Plafkin et al. 1989). Samples are collected from all available habitats using a dip net or kick net, or by hand. To use this method, persons must be able to identify aquatic macroinvertebrate taxa in the field. It can be difficult to identify organisms to the family level without keys or the aid of a laboratory microscope. Benthic macroinvertebrate orders/families collected are listed on a Biosurvey Field Date Sheet, with an estimate of their relative abundance. Relative abundance of other aquatic biota is also recorded, providing additional clues as to impairment.

Impairment may be indicated by the absence of generally pollution-sensitive taxa, dominance of pollution-tolerant taxa, or low abundance and taxa richness. Impairment may also be indicated by an overabundance of slimes or algae and the absence of expected fish species. On the basis of this qualitative biosurvey and observations made on habitat, water quality, and physical characteristics, the investigator determines whether the stream is impaired (Plafkin et al. 1989).

Rapid Bioassessment Protocol II—Benthic Macroinvertebrates (RBP II)

RBP II is a BMP Effectiveness II method that systematically collects and analyzes of major benthic taxa in the field, providing a more intensive assessment than RBP I with relatively little additional time and effort (Plafkin et al. 1989). RBP II can be used to select sites for more intensive evaluation (i.e., RBP III, replicate sampling, etc.) or it can be used in lieu of RBP I as a screening technique. RBP II involves benthic analysis at the family level and uses field sorting and identification.

In the field, the RBP II biosurvey focuses on standardized sampling of benthic macroinvertebrates, supplemented by a cursory field observation of other aquatic biota (periphyton, macrophytes, slimes, and fish). Samples of the macroinvertebrates from comparable habitat at all stations along the stream are collected, plus separate Coarse Particulate Organic Matter (CPOM) samples. RBP II concentrates on riffle/run habitat because it is the most productive habitat available in streams and includes many pollution-sensitive taxa of the scraper and filtering collector functional feeding groups. The CPOM sample provides a measure of the effects of pollution on shredders.

Riffle areas with moderately fast currents and cobble and gravel substrates generally have the most diverse community of benthic macroinvertebrates. The recommended sampling method is to use a kick net to collect from an approximate 1 m^2 area of riffle. Two 1-m^2 riffle samples are collected at each station: one from an area of fast current and one from an area with slower current. The two samples are combined for processing. In streams without riffles, run areas with cobble and gravel substrate can be kick-net sampled. But where riffle/run cobble and gravel habitat does not exist, other types of habitat (submerged boulders, logs, bridge abutments) should be sampled for biota by hand picking. Such habitats are used by scrapers and filtering collectors (Plafkin et al. 1989). Submerged logs and root wads are important habitat in sand-bottomed Coastal Plain streams.

The CPOM is habitat for shredders and is sampled to reveal the abundance of shredders, important in forested stream ecosystems ranging from 1st- to 4th-order streams (Minshall et al. 1985). The absence of shredders is characteristic of unstable headwater streams in disturbed watersheds, which to not retain much CPOM.

The CPOM sample is processed separately from the riffle/run sample and the functional feeding groups are identified (Plafkin et al. 1989). Potential sample sources include leaf packs, shorelines, and other depositional areas where CPOM accumulates. A variety of CPOM types should be collected; samples may be washed in a dip net or a sieve bucket.

Shredders are most abundant when the CPOM is about 50 percent decomposed (Cummins et al. 1989). Avoid collecting fresh or fully decomposed leaf litter to optimize collection of the shredder community. The type of leaf litter will affect the seasonality of shredder abundance.

It is recommended that the riffle/run sample be sorted and enumerated in the field to obtain a 100-count benthic subsample. All invertebrates in the subsample are classified according to functional feeding group. The scraper and filter-collector functional groups are the key indicators in the riffle/run community. Numbers of individuals in these two groups are recorded. All organisms in the subsample are identified as to family or order, enumerated, and recorded along with any observations on abundance of other aquatic biota.

Organisms collected in the CPOM sample are classified as shredders or nonshredders. As with the RBP I method, ability to identify aquatic macroinvertebrates to order and/or family is needed to classify organisms into functional feeding groups in the field. Numbers of individuals representing the Shredder Functional Group, as well as the total number of macroinvertebrates collected in the sample, are recorded.

Eight metrics are used to characterize the biological condition of the study stream versus the reference stream:

1. Taxa Richness
2. Family Biotic Index (pollution tolerance index for families)
3. Ratio of Scrapers/Filter Collectors
4. Ratio of EPT and Chironomid Abundance
5. Percent Contribution of Dominant Family
6. EPT Index
7. Community Loss Index
8. Ratio of Shredders/Total

Each metric is scored based on its percent comparability to a reference stream. Scores are totaled and compared to the total score for the reference stream, providing an evaluation of the biological condition of the study stream.

Caution: Some metrics may be applicable to specific ecoregions and others may not. Metrics 3 and 4 are currently questionable (Karr, personal communication).

Rapid Bioassessment Protocol III—Benthic Macroinvertebrates (RBP III)

RBP III is a more rigorous bioassessment technique than RBP II, involving systematic field collection and subsequent lab analysis, allowing detection of more subtle degrees of impairment (Plafkin et al. 1989). RBP III can be used in BMP Effectiveness III and IV assessments. Invertebrate samples are analyzed in more detail for RBP III, which requires taxonomic identification to genera. Samples must be processed at laboratories specifically designed for such purposes. Use of these lab services is cost-efficient and also ensures adequate quality control. In addition, this protocol provides a basis for monitoring trends or cumulative impacts over time.

Sample collection for RBP III is essentially the same as for RBP II. The riffle/run habitat is the site for sampling scrapers and filtering collectors, supplemented by CPOM samples to collect shredders. Again, submerged boulders, logs, bridge abutments, and pilings are hand picked if riffle/run habitat is absent.

Subsamples of 100 are usually sufficient, but subsample sizes can be 200-300 specimens. All benthic macroinvertebrates in the subsample should be identified to genus or species, if possible, and then classified by functional feeding groups.

The data are analyzed is performed as follows. Using the raw benthic data, a numerical value is calculated for each metric. Calculated values are then compared to values from reference streams or segments on the same stream. Each metric is then assigned a score according to the comparability (percent similarity) of calculated and reference values. Scores for the eight metrics are totaled and compared to the total score for the reference stream. The percent comparison between the total scores provides a final evaluation of biological condition.

The eight metrics are the same as in RBP II.

The study stream can now be classified into one of four biological conditions:

1. Non-Impaired—Comparable to the best situation to be expected within an ecoregion. Balanced trophic structure. Optimum community structure (composition and dominance) for stream size and habitat quality.

2. Slightly Impaired—Community structure less than expected. Composition (species richness) lower than expected due to loss of some intolerant forms. Percent contribution of tolerant forms increases.

3. Moderately Impaired—Fewer species due to loss of most intolerant forms. Reduction in EPT index.

4. Severely Impaired—Few species present. Organisms present are dominated by one or two taxa. Only tolerant organisms present.

Index of Biotic Integrity—Fish (Rapid Bioassessment Protocol V)

The Index of Biotic Integrity (IBI), a fish-community-assessment approach, was developed by Karr (1981). Substantial additional developmental work and regional testing is behind IBI (Fausch et al. 1990, Karr 1991 and 1993, Karr et al. 1986, Lyons 1992, and Miller et al. 1988). It is a rigorous system that should be used primarily at the BMP Effectiveness III and IV levels, but can be adapted to BMP Effectiveness II.

IBI involves careful, standardized field collection, species identification and enumeration, and community analyses using biological indices or quantification of the biomass, and numbers of key species. An IBI survey yields an objective, discrete measure of the health of the fish community that usually can be completed on site by qualified fish biologists. Data collected can serve to assess whether the beneficial use for the stream is attained, develop biological criteria, prioritize sites for further evaluation, provide a reproducible impact assessment, and assess fish community status and trends. IBI is the system used in Rapid Bioassessment Protocol V (Plafkin et al. 1989).

RBP V involves field evaluation of the same physical, chemical, and habitat characteristics as RBP's I, II, and III, a similar impairment assessment, and a fish community biosurvey. Because they provide critical information for evaluating the cause and source of impairment, these data are essential to RBP V.

Electrofishing, the most widely applicable system for sampling stream habitats, is recommended for use with IBI. The biosurvey data are intended to be representative of the fish community at all station habitats. Each sampling station should be representative of the reach, incorporating at least one (preferably two) riffle(s), run(s), and pool(s) if these habitats are typical of the stream in question. Sampling of most species is best done near shore and cover (macrophytes, boulders, snags, brush). The biosurvey is not an exhaustive inventory, but provides a realistic sample of fishes likely to be encountered in the stream.

Typical sampling stations are 100-200 meters long for small streams and 500-1,000 meters for rivers, but lengths are best determined by pilot studies. The reference station should be long enough to produce 100-1,000 individuals and 80-90 percent of the species expected if the station length is increased by 50 percent. Block nets set at both ends of the reach increase sampling efficiency for large, mobile species sampled in small streams. The RBP V fish community assessment requires that all fish species (not just gamefish) be collected.

A field collection data sheet is completed for each sample. Sample duration and area are recorded in order to determine level of effort. Species may be separated into adults and juveniles by size and color; then total numbers and weights and the incidence of external anomalies, such as tumors, deformities, and damaged fins, are recorded for each group.

The steps in determining the IBI are:

1. Select a site.
2. Identify regional fish fauna.
3. Assign species to trophic, tolerance, and origin guilds.
4. Assess available data for metric suitability and stream-size patterns.
5. Develop scoring criteria from reference streams.
6. Quantitatively sample fish.
7. Record abundance of species, hybrids, and anomalies.
8. Calculate and score metric values (Table 10).
9. Add scores to get IBI score.
10. Make recommendations (Table 11).

Table 10. Metric Scores (IBI) (Karr 1981)

Metric	Scoring Criteria		
	5	3	1
1. Number of native fish species	>67%	33-67%	<33%
2. Number of darter or benthic species	>67%	33-67%	<33%
3. Number of sunfish or pool species	>67%	33-67%	<33%
4. Number of sucker or long-lived species	>67%	33-67%	<33%
5. Number of intolerant species	>67%	33-67%	<33%
6. Proportion of green sunfish or tolerant individuals	<10%	10-25%	>25%
7. Proportion omnivorous individuals	<20%	20-45%	>45%
8. Proportion insectivorous	>45%	20-45%	<20%
9. Proportion top carnivores	>5%	1-5%	<1%
10. Total number of individuals	>67%	33-67%	<33%
11. Proportion hybrids or exotics	0%	0-1%	>1%
12. Proportion with disease/anomalies	<1%	1-5%	>5%

Note: Metrics 1-5 are scored relative to the maximum species richness line.
Metric 10 is drawn from reference site data.

Table 11. Index Score Interpretation (Karr 1981)

IBI	Integrity Class	Characteristics
58-60	Excellent	Comparable to pristine conditions, exceptional assemblage of species.
48-52	Good	Decreased species richness, intolerant species in particular decreased; sensitive species present.
40-44	Fair	Intolerant and sensitive species absent, skewed trophic structure.
28-34	Poor	Top carnivores and many expected species absent or rare; omnivores and tolerant species dominant.
12-22	Very Poor	Few species and individuals present; tolerant species dominant; diseased fish frequent.

Metrics will vary by ecoregion and fisheries. The metrics and key habitat elements need to be verified or validated by ecoregion. The above metrics were developed for midwestern streams and may not be applicable to other ecoregions.

Hankin-Reeves—Fish Population Estimation and Community Composition

The Hankin and Reeves (1988) method can be used at the BMP III and IV levels. Visual fish surveys rely on the same premise as visual habitat surveys: if divers count a consistent fraction of the fish present, then there will be a strong correlation between visual estimates and the "true" number of fish. Divers observe the species, number, and usually the size classes of fish in sample habitat units from an entire reach. Like the habitat survey, wherein visual estimates of surface area of habitat units are calibrated by periodic measurements, visual estimates of fish populations are calibrated, but by a more accurate method (such as multi-pass depletion electrofishing) in a fraction of the sampled habitat units.

If a Hankin and Reeves habitat survey was done earlier, that information should be used. If not, the team selects a classification system for identifying habitat types (e.g., Bisson et al. 1981) and stratifies the study area into reaches based on gradient, confluence of same-order channels, or other distinctive feature. The team then decides how many units of each habitat type that will be sampled. The fraction of units sampled need not be the same for every habitat type and should depend on the objectives of the survey. As a general rule, a higher percentage of units should be sampled in habitat types that are preferred by the species of interest. For example, if the species of interest prefers pools, the surveyors might sample 25 percent of the pools and only 10 percent of the riffles and cascades in a study area. However, no habitat type should be eliminated from sampling just because surveyors do not expect fish to be present. There is only one way to confirm the presence or absence of fish in a habitat type: *sample it.*

The next step is to select the percentage of units to be sampled for calibration of diver counts. As a "rule of thumb," at least 10 percent of the units sampled by divers should be sampled by a more accurate method of counting fish (typically multiple-pass depletion electrofishing). The surveyors randomly select the starting unit and interval between units designated for diver calibration.

Population estimates and confidence intervals for species and age-classes in a habitat type are calculated according to Hankin and Reeves (1988) (2 divers count fish in each habitat unit) or Dolloff et al. (1993) (single diver counts fish in each habitat unit).

Using the Hankin and Reeves method on both reference and study streams will allow comparing fish abundance and functional feeding groups between streams. This will reveal major differences in total fish abundance and community composition.

Izaak Walton League—Benthic Macroinvertebrates

The Izaak Walton League (1990) method is a qualitative assessment of benthic invertebrates, coupled with empirical observations of water quality, habitat, fish and land use. This methodology is best used for evaluating invertebrates at the BMP Effectiveness I level. Invertebrates are sampled in three riffles at depths of 3-6 inches and in a 3-foot-square area with a kick net. The invertebrates are

washed into a white pan for identification to order and suborder. Then they are classified as pollution sensitive, somewhat tolerant, or tolerant, using a simple chart with taxonomic drawings of invertebrates that are keyed to pollution tolerance.

The numbers of individuals in each order found in the 3-foot square are reduced to a code: A = 1-9, B = 10-99, and C = 100+ . The appropriate letter is assigned to each order and suborder, and recorded on the form below.

MACROINVERTEBRATE COUNT

	Water Quality Indicator	
Good	**Fair**	**Poor**
___ Caddisfly	___ Beetle larvae	___ Aquatic worms
___ Dobsonfly	___ Clams	___ Blackfly
___ Mayfly	___ Crane fly	___ Leeches
___ Stonefly	___ Crayfish	___ Midges
___ Other snails	___ Damselfly	___ Pouch snails
___ Riffle beetles	___ Dragonfly	
___ Water penny	___ Scuds	
	___ Sowbugs	
	___ Antherix	
___ # of letters times 3 =	___ # of letters times 2 =	___ # of letters times 1 =
___ Index value+	___ Index value +	___ Index value

Then the number of letters in each column is multiplied by the indicated index. The three index values are added to get a total index value, which is then classified as: Excellent (> 22), Good (17-22), Fair (11-16), or Poor (< 11). This rating of excellent, good, fair, or poor is a qualitative assessment of water quality as reflected in the benthic community. This method is usually used to monitor the condition or trends in one stream, or above and below a management area.

To evaluate the effectiveness of BMP's, study streams must be compared to reference streams. If the ecoregion has been impacted by prior land use, the reference stream may be assessed as only fair. Without the reference assessment, BMP's near a study stream rated as fair might be erroneously judged as ineffective when indeed they could be effective.

Because invertebrate numbers are lumped into only three classes, study and reference streams may receive the same qualitative rating, when in fact there may be significant differences in the number and taxa of invertebrates. So the simple rating of excellent, good, fair, or poor can be misleading as can comparing total index values between study and reference streams. However, by comparing the occurrence of orders of invertebrates and of numbers of individuals between study and reference stream, qualitative assessments of BMP effectiveness can be made with the Izaak Walton League method.

Vermont Guide to Macroinvertebrate Sampling

The River Watch Network in Vermont (1992) has developed a guide for macroinvertebrate sampling which was adapted from Rapid Bioassessment Protocols (Plafkin 1989) for local conditions. It uses two levels of data analysis.

Level 1 classifies invertebrates into major groups (mostly orders) to reveal the basic composition of the macroinvertebrate community as well as serious changes resulting from changes in the stream ecosystem. The metrics used in this analysis are taxa richness, organism density per sample, EPT richness (orders), and EPT-to-Chironomidae ratio. This level of analysis is appropriate for BMP Effectiveness I and II.

Level 2 identifies macroinvertebrates to the family level, refines one data analysis procedure in Level 1, and adds two others, all of which reveal more subtle differences among sites. The refined analysis is EPT richness at the family level, and the added analyses are the modified family biotic index (based upon Hilsenhoff tolerance indices to pollution) and Pinkham and Pearson community similarity index. This is a BMP Effectiveness III method.

The stream conditions measured or estimated are: water temperature, air temperature, average width, average depth, relative flow compared to average annual flow, sediment deposits, smell of water, water color, algal growth, channelization, upstream dam, point discharges, stream bank characteristics, substrate composition, embeddedness, and overhead canopy.

Basin Area Stream Survey (BASS)

Clingenpeel and Cochran (1992) describe a process for evaluating the physical, chemical, and biological indicators of water quality on the Ouachita National Forest in Arkansas called the BASS methods. BASS elaborates on methods developed by Pfankuch (1975), Hankin (1984), Bisson et al. (1982), Ebert et al. (1987 and 1991), and Filipek et al. (forthcoming). It includes habitat typing and geographical-information-system mapping, evaluating habitat and water quality, and sampling fish and benthic invertebrates. BASS uses reference and study streams within ecoregions. It is a BMP Effectiveness III method.

The biological inventory is based upon a 10-percent sample of habitat types along study and reference streams. For example, if 27 main channel pools are identified within a stream, three of these pools are sampled.

Fish populations in each reach are isolated with block nets before sampling by electrofishing equipment. Fish in the sample area are collected using at least two but preferably three or more passes for depletion sampling. With each pass, shocked fish are collected, reducing the number of fish remaining in the stream. Depletion sampling will collect most of the fish in the stream segment. Electrofishing passes cover the entire reach from the downstream net up, with equal effort applied on each pass. The downstream net is checked for fish after each pass and captured fish are included with the sample for that pass. Each pass comprises a sample and the fish are placed in separate containers. Game, endangered, threatened, and sensitive species are identified, measured, and

weighed in the field and returned to the stream. The remaining fish are preserved in 10-percent formalin and labeled to allow follow-up analysis, if needed. The preserved fish are identified, sorted, and measured in a laboratory. Population of each species is estimated using the maximum-likelihood method of VanDeventer and Platts (1990).

Macroinvertebrate samples are collected in the same reaches as sampled for fish, using a 5-minute kick-net sample. Collecting is done by shuffling or kicking the substrate directly above the kick net. All microhabitats (woody debris, leaf packs, etc.) within the reach are sampled. In addition, a 5-minute wash sample of the substrate is taken. A dip net is placed in the stream and individual cobbles above it are scrubbed with a soft-bristle brush into the net. The wash sample is combined with the kick-net sample. Large organic matter and leaves are washed and removed from the sample. The invertebrate samples are preserved in 70-percent ethanol and labeled. Identification and sorting are done in a laboratory.

Ohio EPA Macroinvertebrate Sampling

As an example of BMP Effectiveness III and IV methods being developed by State water-quality agencies, Ohio EPA (1987 and 1989) developed a method for assessing benthic macroinvertebrates using artificial substrates supplemented with a qualitative sample of the natural substrate. The primary sampling equipment used for collecting benthic macroinvertebrates is the modified Hester-Dendy, multiple-plate, artificial-substrate sampler, which is placed in the stream for a period of time to allow invertebrates to colonize it. The sampler is constructed of 1/8-inch tempered hardboard cut into 3-inch-square plates and 1-inch square spacers, which are held together at the center by a 1/4-inch eye-bolt. Eight plates and twelve spacers are used for each sampler. The plates and spacers are arranged on the eyebolt so that there are three single spaces, three double spaces and one triple space between plates.

Before the samplers are placed in the stream, they are tied to a concrete block, which anchors them in place and prevents the plates from touching the natural substrate. In water deeper than 4 feet, a float is attached to the samplers to keep them within 4 feet of the surface. Whenever possible, the samplers are placed in runs rather than pools or riffles and stations are established in as similar an ecological situation as possible. The samplers are exposed for 6 weeks.

The samplers are retrieved by cutting them from the block and placing them in 1-quart plastic containers while still submerged. Care is taken to avoid disturbing the samplers and thereby dislodging any organisms. Enough formalin is added to each container to approximate a 10-percent solution.

Qualitative samples of macroinvertebrates inhabiting natural substrates are collected at the time of sampler retrieval. In shallow water, samples are taken in a stream segment covering all available habitats in the vicinity of the samplers. Samples are collected using triangular-ring-frame, 30-mesh dip nets and hand picking with forceps. The sampling continues until, by visual examination, no new taxa are being taken.

Where quantitative biological samples are collected from the natural substrate, the collector stands on the downstream side of a Surber square-foot sampler (30-mesh netting) sampler and works the substrate using a 2-tined hand cultivator. Large rocks are gently scrubbed with a brush. The material collected is placed in sealed containers, preserved in 10-percent formalin, and transported to the laboratory. Three to five Surber samples are taken at each site.

Laboratory methods for quantitative sampling include taking composite samples with five multi-plate samplers to be used in station evaluations for routine monitoring. For litigation and other needs, replicate sampling and larger composite samples can be used. The Ohio EPA has established a laboratory protocol for identifying and enumerating invertebrates, generally to the genus or species level. For the qualitative sample, Ephemeroptera, Plecoptera, and Trichoptera are enumerated.

The principal measure of overall macroinvertebrate community condition is the Invertebrate Community Index (ICI). The ICI is a modification of the Index of Biotic Integrity (IBI) for fish developed by Karr (1981). It consists of ten structural community metrics, each with four scoring categories of 6, 4, 2, and 0 points (Table 12).

The point system evaluates a sample against a database of 247 relatively undisturbed reference sites throughout Ohio. Six points are given if a metric has a value comparable to exceptional stream communities, 4 points for those characteristic of more typical good communities, 2 points for metric values slightly deviating from the expected range of good values, and 0 points for metric values strongly deviating from the expected range of good values. A community-similarity index and rank correlation are used to test similarity between study streams and reference streams.

Table 12. Invertebrate Community Index (ICI) metrics and scoring criteria based on macroinvertebrate community data from 247 reference sites throughout Ohio

Metric	Scoring Criteria 0 2 4 6
1. Number of Taxa 2. Total Number of Mayfly Taxa 3. Total Number of Caddisfly Taxa 4. Total Number of Diptera Taxa 5. Percent Mayflies 6. Percent Caddisflies 7. Percent Tribe Tanytarsini Midges 8. Percent other Dipterans and Noninsects 9. Percent Tolerant Organisms 10. Total Number of Qualitative Ephemeroptera, Plecoptera, and Trichoptera (EPT) Taxa	Scoring of each metric varies with drainage area (Ohio 1987).

Ohio EPA Fish Sampling

The Ohio EPA samples fish using several methods to calculate the Index of Biotic Integrity (IBI) and the Modified Index of Wellness (IBW) from which aquatic life use attainment is partially judged. This is an example of a BMP Effectiveness III method developed by State water-quality agencies for fish.

Fish sampling sites are selected according to several factors including, but not limited to, the following:

1. location of point source dischargers
2. stream use designation evaluation
3. location of physical habitat features (e.g., dams, changes in geology, changes in stream order, presence of a stream confluence, etc.)
4. location of nonpoint sources of pollution
5. variations in macrohabitat
6. proximity to ecoregion boundaries

Each study area is allocated a number of biological sampling sites allocated based on the number and complexity of the priority issues requiring field evaluation. The principle objectives of each survey determine where sampling sites will be located. Placement of sampling sites is influenced by practical access and resource constraints. Generally, sites are located upstream from all pollution sources to determine the background condition for the study area. Should the upstream portion of the stream be impacted, an alternate site may be chosen on an adjacent stream with similar watershed characteristics. Reference sites within the same ecoregion may also be used. The role of upstream sites is not necessarily to provide a biological performance level against which downstream sites are compared since the ecoregion biocriteria serve this purpose for aquatic life use designations. Upstream sites are, however, important in defining any site or watershed specific background conditions that might temporarily or permanently influence eventual aquatic life use attainment in the downstream reaches. The most typical habitat available is selected for sampling within a segment in an effort to represent the current potential of that segment. An attempt should be made to sample similar macrohabitats at all sampling sites within the study area.

The Ohio EPA uses mainly pulsed, direct-current electrofishing to measure fish abundance and distribution. Boat mounted electrofishers are used in large streams and rivers; backpack electrofishers are used in wadable streams.

Fish are best sampled between mid-June and early October, when stream and river flows are generally low, pollution stresses are potentially the greatest, and the fish community is most stable and sedentary. Since the ability of the netter to see stunned fish is critical, sampling should take place only during periods of "normal" water clarity and flow. "Normal" water conditions in Ohio occur during below annual average river discharge levels when the surface of the water generally has a "placid" appearance. Abnormal turbidity is to be avoided as are high flow and rapid current. All these reduce sampling efficiency and may disqualify data for calculating the modified IWB and IBI.

Captured fish are immediately placed in an on-board livewell for later processing when sampling each site is complete or when the livewell is full. Water in the livewell should be changed regularly to minimize mortality of captured fish. Fish are released immediately after they are identified as to species, examined for external anomalies, and, if necessary, weighed. Most captured fish can be identified to species in the field; however, if there is any uncertainty, the fish must be preserved for later laboratory identification. The data are used to develop IBI and IWB scores for comparing study streams with reference conditions.

Snorkel Methods for Fish

Thurow (1991) describes a snorkeling method, suited for BMP Effectiveness II and III assessments, to visually determine fish species, age, and size. The method is described for evaluating salmonids. Observations should be made after the stream temperature exceeds 9 °C. Divers must be trained to judge species, age, and size of fish. Snorkelers enter the water downstream of the habitat to be surveyed and proceed slowly upstream. Data are recorded on a PVC sleeve or slate with a lead pencil.

Streamwalk

The USEPA Region 10 has begun a citizens monitoring program called Streamwalk. Used by a professional, it can yield meaningful results at the BMP Effectiveness I level. The empirical observations include stream characteristics, stream-bottom type, width of streamside corridor, streamside vegetation, canopy cover, presence of large woody debris, adjacent land use, stream-bank conditions, and others. Streamwalk is used to identify potential problem areas and monitor trends.

Streamwalk II

In Idaho, Streamwalk has been expanded to be more rigorous and more demanding of participants, and thus has become Streamwalk II (Rabe 1992). Streamwalk II assesses habitat, chemical and physical conditions, and biology. Conditions in study streams are compared to reference-stream conditions. This method is applicable to the BMP Effectiveness II level of monitoring.

The habitat assessment procedure was adapted from the Rapid Bioassessment Protocols (Plafkin 1989) and information provided by the State DEQ and university personnel from Idaho. Variables are weighted to emphasize biological criteria. After the study reach has been surveyed, ratings are totalled and compared to a reference. The study reach score is divided by the reference score to calculate a percent comparability score for each variable and total score. Reference streams represent the best habitat conditions attainable for a region.

The primary habitat variables (bottom substrate, embeddedness, and velocity/depth) are evaluated in a riffle-pool sequence and scored from 0-20 points. Secondary variables consist of wetted channel shape, pool/riffle ratio, and channel alteration (0-15 points). Tertiary variables include lower-bank stability, bank vegetation protection, canopy cover, and width of riparian zone (0-10 points).

The chemical and physical quality of water is analyzed and a Water Quality Index determined. Ten variables are assessed: dissolved oxygen, pH, alkalinity, and conductivity are analyzed in the field; nitrates, phosphates, total solids, turbidity, fecal coliforms, biochemical oxygen demand, and pH are analyzed in a lab.

Biomonitoring includes systematic sampling of habitat types with a variety of sampling gear. Comparisons are made within and between study and reference streams to detect the presence of macroinvertebrates that are sensitive or tolerant to water-quality impacts and to determine taxa richness. As many taxa as possible are collected in a reasonable period of time. The habitat types sampled and the associated organisms are recorded. Habitat types include riffles, pools, aquatic plants, coarse particulate material, and soft sediments.

The invertebrate samples are processed and the following biological metrics are developed:

1. Species richness—the total number of different taxa present in the sample.
2. EPT index—the total number of Ephemeroptera (mayflies), Plecoptera (stoneflies), and Trichoptera (caddisworms) found in the sample. Insects comprising these three orders are generally sensitive to pollution. A high EPT index is associated with clean water and unimpacted habitat.
3. Biotic index—a value given to an organism depending upon its tolerance to pollution. Low values indicate organisms sensitive to pollution. High values indicate organisms more tolerant and less sensitive to stress. The Hilsenoff index is used.
4. Dominance—the relative number of individuals belonging to one or two species in the community. The dominance of one or two species usually relates to an impacted habitat or poor water quality.
5. Functional feeding groups—functions or activities are associated with different macroinvertebrates in a stream. If several different types are present, then this condition reflects a healthy ecosystem as compared to a stream with only a few functional types.
6. Index of similarity—the similarity of taxa between study and reference streams.

Hilsenhoff Biotic Index

Water-quality condition can be determined simply using benthic macroinvertebrate sampling and the Hilsenhoff Biotic Index (1987). The following was developed in Idaho and represents a BMP Effectiveness I technique.

The relative health of a stream can be estimated by collecting a macroinvertebrate sample from a stream riffle section, identifying the basic composition of invertebrate taxa in the sample, and averaging Hilsenhoff tolerance values over all specimens examined. The procedure is as follows:

1. Obtain a 3' x 3' Section of fine mesh screen material and attach two 2.5' lengths of 1" diameter wood dowels to opposite edges of the screen to make the kick net.

Table 14. Methods used for the standardized qualitative collection techniques (Lenat 1988)

Sampler	Habitat	Number of Samples	Type
Coarse-mesh[a]			
Kick net (high current with structure[b])	Usually riffles, also snags in sandy streams	2	Single, disturbance
Dip net (low current with structure)	Usually banks	3	Composite, disturbance
Wash bucket (leaves)	Leaf packs	1	Composite, wash
Fine-mesh[c]			
Plastic basin + PCV cylinder (aufwuchs)	Rocks, logs	2	Composite, wash
Nitex bag (sand)	Sand	1	Composite (3), disturbance
Visual collections	Large rocks & logs	1	Composite

[a] 0.5-1.0 mm mesh.

[b] Substrate (rocks, logs, roots) that offers protection from scour or place of attachment.

[c] 0.3 mm Nitex mesh.

North Carolina Rapid Bioassessment Method (Four Sample)

North Carolina developed a four-sample technique as a rapid bioassessment method (Lenat 1988) to satisfy a need for a more rapid survey technique than their standard qualitative collection method (10-sample). This techniques is applicable to BMP Effectiveness II and III levels. A modification of their 10-sample technique, the four samples consist of: a kick-net collection, a sweep-net collection (bank area), a leaf pack, and picking insects from large rocks and logs. This method focuses on EPT taxa richness. Eaton and Lenat (1991) report a strong correlation between EPT taxa per the rapid method and the standard method, and that the rapid EPT method does nearly as good a job of predicting water quality as standard qualitative methods. Sampling and processing take half as long as standard qualitative methods.

They caution using the rapid method if EPT taxa richness is low, such as in very small streams, some swampy coastal areas, and areas where the "control" site is already severely polluted (Eaton and Lenat 1991). In these situations, the 10-sample method should be used. They suggest that EPT collections be supplemented by notes on the abundance of obvious pollution-indicator groups.

Aquatic Vegetation and Zooplankton Monitoring Methods

Aquatic vegetation and zooplankton are important elements of stream ecosystems and can be used as indicators of stream health. EPA's draft guidance "Biological Criteria: Technical Guidance for Streams" (Barbour et al. 1991) discusses periphyton, macrophytes, and zooplankton as possible biological criteria for inclusion in State water-quality standards. The following is quoted from their draft guidance (for references cited in this excerpt, see Barbour et al. 1991).

> *Periphyton: The broad organismal assemblage known as periphyton is composed of benthic algae, bacteria, their secretions, associated detritus and various species of microinvertebrates (Laberti and Moore 1984). Periphyton is an important energy base for many lotic situations (Steinman and Parker 1990; Minshall 1978; Dudley et al. 1986) serving as the primary nutrient source for many stream organisms (Lamberti and Moore 1984). The capacity of these benthic assemblages to colonize and increase in biomass is influenced by variability in stream channel geomorphology, flow rates, herbivore grazing pressure, light intensity, seasonality, and random processes (Stevenson 1990; Grimm and Fisher 1989; Coleman and Dahm 1990; Poff et al. 1990; Korte and Blinn 1983; Hamilton and Duthie 1984; Lamberit et al. 1987; Steinman and McIntire 1986, 1987; Steinman et al. 1987). The dynamic position of the periphyton assemblage within most stream ecosystems makes it a prime for consideration as a bioassessment/biosurvey target. More specific advantages given by Plafkin et al. (1989) are that:*
>
> - *rapid algal reproduction rates and short life cycles make them valuable as indicators of short-term impacts;*
> - *as the periphyton assemblage is a function of primary production, physical and chemical factors have direct effect;*
> - *sampling methods are straightforward and sampling effort is easily quantified and standardized;*
> - *methods have been relatively standardized for functional and nontaxonomic characteristics of periphyton communities such as biomass and chlorophyll measurements; and*
> - *algal components of periphyton are sensitive to some pollutants to which other organisms may be relatively tolerant.*
>
> *Macrophytes: Both emergent and submergent vegetation provide numerous features to lotic situations which enable them to support healthy, dynamic, biological communities (Miller et al. 1989; Campbell and Clark 1983; Hurley 1990). Spatio-temporal distributional characteristics and some understanding of the effecting environmental conditions of macrophytes (Hynes 1970) allow their use in bioassessment strategies. Hynes (1970) and Westlake (1975) discuss differences in lotic macrophyte assemblages based on habitat factors such as water hardness, pH, gradient, and propensity for siltation. Other workers have emphasized the capacity of macrophytes to influence habitat structure (Miller et al. 1989; Gregg and Rose 1982; McDermid and Naiman 1983; Pandit 1984; Carpenter and Lodge 1986), water chemistry, nutrient*

cycling, and macroinvertebrate colonization (Miller et al. 1989; McDermid and Naiman 1983). Seddon (1972), Westlake (1975) and Pandit (1984) point out the utility of macrophytes as an indicator assemblage in lotic situations.

Aquatic macrophytes are important as a food source for birds and mammals. Fassett (1957) lists 36 species of waterfowl, nine marshbirds, four shorebirds and nine upland game birds feeding on these plants. He also lists beaver, deer, moose, muskrat, and porcupines as aquatic macrophyte herbivores. From these and other considerations, a number of advantages are apparent for use of macrophytes in bioassessment programs:

- *taxonomy to the generic level is relatively straightforward;*
- *establishment of populations in a specific habitat may be somewhat dependent on local environmental conditions; thus, there is potential as site-specific indicators;*
- *specific microhabitat structure may not limit germination; thus, there is potential for high population density;*
- *growth patterns of individual plants are directly influenced by herbivore activity;*
- *longevity, distribution, and rate of population growth may directly reflect prevailing conditions.*

Zooplankton: These organisms are generally considered the second link (primary consumers) in aquatic food chains, although certain common limnetic taxa such as Cyclops species prey upon zooplankton and are regarded as secondary consumers. Zooplankton that are primary consumers feed on phytoplankton, bacteria, protozoa, and organic detritus and are in turn utilized by aquatic macroinvertebrates and fish (secondary and tertiary consumers). Many fish species at some stage of development use zooplankton as a food source. Availability of sufficient quantities of zooplankton at critical periods may affect survival of fry and fingerling stages of fish species.

Alterations of the environment, such as increases in water temperature, turbidity, or changes in the water circulation pattern could influence densities and species composition of indigenous zooplankton populations, thereby affecting zooplankton-dependent organisms at higher trophic levels.

Advantages in use of zooplankton in biocriteria are the following:

- *high diversity,*
- *can attain high abundance levels in low gradient river systems,*
- *likely to encompass broad range of sensitivities,*
- *brief life cycles compared to higher trophic levels, and thus are responsive to short-term perturbations,*
- *field sampling methods are cost-effective and straight forward.*

Periphyton are algae, fungi, bacteria, protozoa, and associated organic matter attached to channel substrates (Klemm and Lazorchak, 1993). Periphyton are useful indicators of environmental conditions because they respond rapidly, are non-mobile, and are sensitive to a number of anthropogenic disturbances, including contamination by nutrients, metals, herbicides,

hydrocarbons, and acids. Periphyton represent another trophic level, exhibit a different range of sensitivities and will often indicate effects only indirectly observed in the benthic macroinvertebrates and fish communities (Plafkin et al. 1989, Hayslip 1993). Hayslip (1993) recommends considering the following periphyton analyses:

1. *Chlorophyll a*
2. *Ash free dry mass (AFDM)*
3. *Diversity indices*
4. *Taxa richness*
5. *Indicator species*

Klemm and Lazorchak (1993) describe periphyton sampling methods, laboratory and quality assurance/quality control procedures.

Periphyton and Macrophyte Monitoring Methods by Monitoring Level

As with all elements being monitored, comparisons are made between reference and study stream periphyton and macrophyte communities. Monitoring methods are listed below, followed by abstracts of the various recommended methods for each monitoring level.

BMP Effectiveness I	Page
Protocol I—Periphyton	150
Protocol I—Aquatic Macrophytes	150
Montana Periphyton Protocol I	151
BMP Effectiveness II	
Protocol II—Periphyton	150
Protocol II—Aquatic Macrophytes	150
Montana Periphyton Protocol I or II	151/152
BMP Effectiveness III	
Protocol II or III—Periphyton	150
Protocol III—Aquatic Macrophytes	150
Montana Periphyton Protocol II	152
BMP Effectiveness IV	
Protocol III—Periphyton	150
Protocol III—Aquatic Macrophytes	151

The methods for evaluating periphyton and macrophytes are in the early stages of development for inclusion in the Rapid Bioassessment Protocols (personal communication from Brian H. Hill and James M. Lazorchak, USEPA Environmental Monitoring Systems Lab, Cincinnati). Their suggested RBP approach for aquatic plants is as follows:

Protocol I—Periphyton

Protocol I is a cursory survey for the presence and relative abundance of periphyton, including attached filamentous algae and blue-green algae. Presence of a periphyton community is determined by visual observation (growth on substrate, texture, color) and feeling the substrate for "sliminess". Filamentous algae are visible "streamers" attached to substrates. Blue-green algae are notable only when they are at nuisance levels and are detectable by color and odor. Periphyton, exclusive of filamentous algae and blue-green algae, are scored as being absent or not apparent, rare, common, abundant, or dominant. This method can be used at the BMP Effectiveness I level.

Protocol II—Periphyton

Protocol II requires qualitative sampling (e.g., scraping from rocks and collecting of fine sediments in depositional areas) of representative substrates for analysis in the laboratory. Laboratory analysis involves viewing an aliquot of suspended periphyton under high magnification (400-1000x) to determine the proportions of diatoms, blue-green algae, and other algae. Only gross morphology is used to separate diatoms and blue-green algae from the other algal taxon. This is a BMP Effectiveness II method.

Protocol III—Periphyton

At a BMP Effectiveness III or IV level, Protocol III requires a minimum of three quantitative samples (e.g., scraping a given area) of the periphyton community. This number needs to multiplied by the number of replicate samples that will be taken at each site. Samples are analyzed for species composition density, chlorophyll concentration, and biomass (ash free dry mass, AFDM). One of the three scrapings is sub-sampled and processed according to standard procedures to allow for species determination and counts of organisms present. The second scraping is filtered and analyzed for chlorophyll concentration. The third is filtered, dried, weighed, and ashed to determine AFDM.

Protocol I—Aquatic Macrophytes

Protocol I includes qualitative estimates of aquatic macrophytes and the proportion of the community composed of submerged, floating-leaved, and emergent life forms. In addition, the percent of the stream occupied by these macrophytes is estimated. BMP Effectiveness I can use this method. A more advanced Protocol I involves collecting and field-identifying (to the lowest possible taxon) aquatic macrophytes present.

Protocol II—Aquatic Macrophytes

Protocol II involves qualitative (line or belt transects) to estimate the relative abundance of species present. Samples may be sorted, identified, and counted in the field or returned to the lab for processing. BMP Effectiveness II can use this method.

Protocol III—Aquatic Macrophytes

Protocol III is based on quantitative sampling. Samples are collected to determine relative abundance and biomass by species. Samples may be sorted, identified, counted, and weighed in the field or returned to the laboratory for processing. Level III sampling on successive dates (e.g., monthly during the growing season) reveals aquatic macrophyte productivity. RBP III is applicable to BMP Effectiveness III and IV levels.

Montana Periphyton Protocol I

The Montana Water Quality Bureau has developed a manual that presents guidelines for using the composition and structure of periphyton communities to assess biological integrity and impairment of aquatic life in Montana streams (Bahls 1993). The manual recommends: (1) methods for collecting, processing, and analyzing periphyton samples; (2) measurements and metrics for evaluating periphyton communities; and (3) biocriteria and protocols for assessing biological integrity and aquatic-life impairment. The document also discusses variability in metrics.

The manual describes two protocols: Protocol I compares study-stream impairment compared to ecoregion reference conditions; Protocol II assesses impairment based upon comparisons with upstream or side stream conditions. Both protocols distinguish among four levels of biological integrity and impairment of aquatic life.

Summer (June 21 to September 21) is the preferred time for collecting periphyton from Montana streams and is the index period for establishing ecoregion reference conditions. In Montana, periphyton diversity peaks in summer and early fall. If sampling must be done outside this period, Protocol II should be used.

Periphyton are collected from natural substrates (rocks, logs, moss, mud) in proportion to the abundance of these substrates at a given site and then pooled in a common container. Microalgae are collected by scraping the entire surface of several rocks of different sizes selected at random. Portions of macroalgae (algae growing in large filaments or colonies visible to the naked eye) are collected in proportion to their abundance at the site and added to the common container. Sampling is usually concentrated in riffles, but other macrohabitats (pools and runs) are also sampled if they support algal growth.

A two step analysis is used: (1) a ranking of non-diatom (soft-bodied) genera by relative volume (biomass), and (2) relative abundance of diatom species. Soft-bodied algae are classified by (1) dominant phylum, (2) indicator taxa, and (3) number of genera. Dominance of blue-green algae may reflect a low level of inorganic nitrogen in mountain streams, while green algae may indicate nitrogen enrichment. Certain genera of non-diatom algae can be used as indicators of different levels and causes of pollution. The number of non-diatom genera is inversely proportional to the degree of pollution. However, in infertile mountain streams a small increase in nutrients can increase the number of genera.

For diatom species, four metrics are used: (1) diversity index, (2) pollution index, (3) siltation index, and (4) similarity index. The Shannon diversity index incorporates dominance and species richness and is sensitive to water-quality impacts. Diatoms are rated as tolerant, less tolerant, and sensitive to pollution. The siltation index is based on the abundance of species that are sensitive to sedimentation.

Protocol I is used to compare study streams with ecoregion reference conditions and can only be used during the summer. This protocol would be appropriate for BMP Effectiveness I and II.

Montana Periphyton Protocol II

This protocol compares metric values for a study site to those for upstream reaches or side-stream reference sites (Bahls 1993). The reference site must have the same stream order as the study reach. This protocol uses the same three diatom indexes as Protocol I plus the percent-similarity index. Protocol II is more sensitive than Protocol I because it compares study streams with local reference conditions. And, unlike Protocol I, it can be applied year around. The siltation rating method used in Protocol II puts a greater penalty on sediment increases at the lower end of the siltation index. This method applies to BMP Effectiveness II and III.

Deciding on BMP Effectiveness: Some Case Histories

The monitoring methods just described can be applied in various ways. The examples reviewed below, real and hypothetical, illustrate how some of these methods can be used. These examples demonstrate multi-factor monitoring to collect, analyze, and present data and information for decisions on whether BMP's are effective in protecting the physical, chemical, and biological integrity of streams.

Implementation and Effectiveness Monitoring in South Carolina

T. Adams (1992) monitored the implementation and effectiveness of harvesting BMP's in South Carolina using RBP III at the BMP Effectiveness III level. The study involved 27 harvest areas associated with perennial streams. He rated BMP implementation as adequate or inadequate, traced soil movement from its source to the stream, and identified sources of sediment found in the stream. Reference stream segments above the harvest area or in nearby watersheds were carefully selected to represent least-impaired streams. Similar stream segments below the harvest area were located for sampling. He used Rapid Bioassessment Protocol III (RBP III) to sample and identify benthic invertebrates to the species level and to assess the biological integrity of the reference and study streams. The RBP habitat evaluation was used to compare habitat quality with the reference segments. The biological and habitat metrics were analyzed to determine which ones were most important for the stream/ecosystems being monitored.

The habitat and biological scores for stream segments below logged areas were divided by reference scores to obtain the percent of reference and plotted (Figure 24). Based upon analysis of data, biological condition in excess of 75% of the reference condition was judged as non-impacted. Habitat was judged as non-impaired if it exceeded 77% of the reference. These thresholds closely correspond to RBP guidance (EPA 1989).

Of the 16 logged areas with adequate BMP implementation, 13 stream segments below have unimpacted habitat and biological conditions, 2 have unimpacted habitat but impacted biological condition, and 1 has impacted habitat and unimpacted biological condition.

Stream segments below 10 of 11 logged areas without adequate BMP implementation generally had impacted habitat and/or biological conditions (Figure 24). However, one logged area that had a road crossing a stream with a very small culvert appeared unimpacted because a high-flow event washed out the road fill and apparently carried the sediment far below the study reach.

Harvest areas rated as having inadequate BMP compliance had been logged under wet conditions and had inadequate streamside management zones, poorly designed skid trail stream crossings, poorly located skid trails, inadequate sediment control from roads, poor road crossings of streams, excessive rutting, etc. Stream habitat was impaired primarily by excessive amounts of large woody debris and sediment. The increase in cobble embeddedness, the loss of available macroinvertebrate habitat, and the loss of riparian canopy all adversely

impacted the streams. The health of the invertebrate community increased with the level of BMP compliance. Multi-factor monitoring linked poor logging practices with impaired benthic invertebrate condition and aquatic habitat. In short, the diagnosis is: poor logging impairs water quality. Generally, water quality is adequately protected where the logger did a good to excellent job of implementing BMP's.

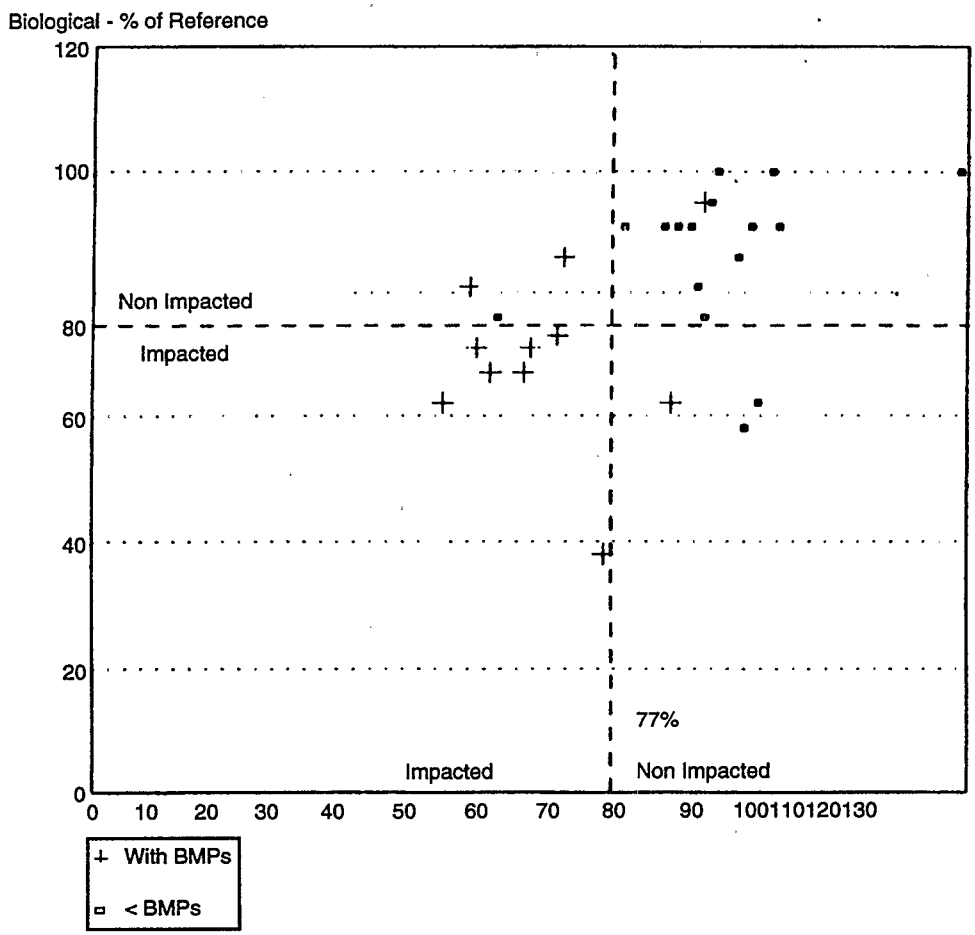

Figure 24. Effectiveness of logging practices in protecting benthic macroinvertebrates and habitat.

A Water-Quality Criteria Approach

Using Adams' (1992) data for the Piedmont region of South Carolina, this example demonstrates how water-quality criteria can be used to evaluate streams at the BMP Effectiveness III level of monitoring. For this example, only biological conditions will be discussed. The Rapid Bioassessment Protocol III for benthic macroinvertebrates was used to determine the biological condition for each reference and study stream. Sixteen reference streams were assessed and the biological-condition scores were plotted (Figure 25). The reference scores for the Piedmont ecoregion ranged from 36 to 42, with an average of 40, which defines a reference condition. About 75 percent of the scores were above 39. A State might therefore set the criterion for biological condition at 39 but not take enforcement action if the stream under investigation fell within the reference condition range.

Stream segments below 11 logging areas with BMP's had biological conditions ranging from 30 to 42 and averaging 37. Nine of these segments had scores within the range of the reference condition. Two had scores of 32 and 30, well below the reference condition. Follow-up field investigation of these two logging areas might find them in general compliance with BMP's, although one or two BMP's might be only partially implemented or lacking, or the BMP's might not be totally effective in those situations.

Four logged areas with poor BMP implementation had scores ranging from 24 to 36, with an average of 32 (Figure 25). The BMP compliance evaluation found several poor logging practices that resulted in organic debris and sediment loading of streams and reduced stream shade. The 36 score falls at the bottom of the reference range, but it is not unexpected for poor logging to meet reference conditions in specific situations. However, three study segments fell well below the reference range, indicating these violated criteria for biological condition.

Multi-Factor Approach/Stream Continuum

For a hypothetical situation at the BMP Effectiveness III level of monitoring, a continuum might be used to evaluate the effectiveness of BMP's. In a reference stream, the index of biological integrity for fish (IBI), embeddedness, and stream temperature are evaluated along the continuum from the headwaters downstream through a fourth order segment (Figure 26). In the figure, the stream flows from left to right. Stream order is on the horizonal axis. Variations in the index of biological integrity, embeddedness, and stream temperature are plotted, trends established, and data analyzed.

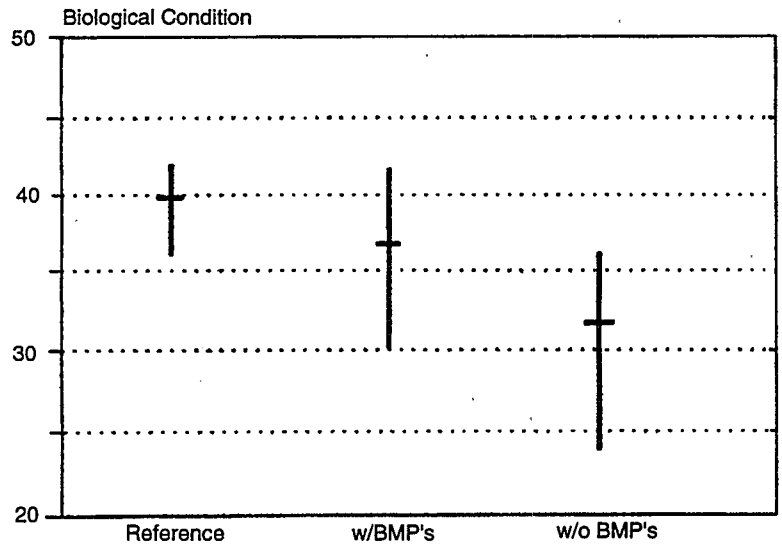

Figure 25. Benthic macroinvertebrate response to logging with and without BMP's, Piedmont, S. Carolina. (Vertical line depicts range in values. Horizonal line represents the mean value.)

The same variables are evaluated for the study stream (Figure 26). The study watershed features logging with and without BMP's. The trends and variation in IBI, embeddedness, and temperature can be compared between study and reference streams and along streams.

Above the logging with BMP's, IBI, embeddedness, and temperature variation and trends in the study stream appear to be very similar to the reference conditions. As the study stream proceeds past the logging with BMP's, IBI decreases slightly, embeddedness is unchanged, and temperature increases slightly—variations and trends similar to those for a comparable segment of the reference stream. On-slope monitoring of the logging found that BMP's had been installed correctly, no sediment reached the stream, and the riparian canopy density was maintained at prescribed levels. The evidence strongly suggests that the BMP's were effective in protecting water quality to this point along the stream.

As the stream proceeds past the logging without BMP's, IBI decreases significantly (Figure 26), stream temperature increases to a near critical level, and embeddedness in the stream increases significantly. Moreover, 75 percent of the riparian canopy was removed by the logging. Soil erosion from skid trails, logging decks, and roads is traced through the riparian area to the stream. All three in-stream responses can be linked to the poor logging and are clear departures from reference conditions and natural variation, leading to a diagnosis that the poor logging has impaired water quality. The next step may be comparing the impacts with State water-quality criteria (if available) to determine if violations have occurred.

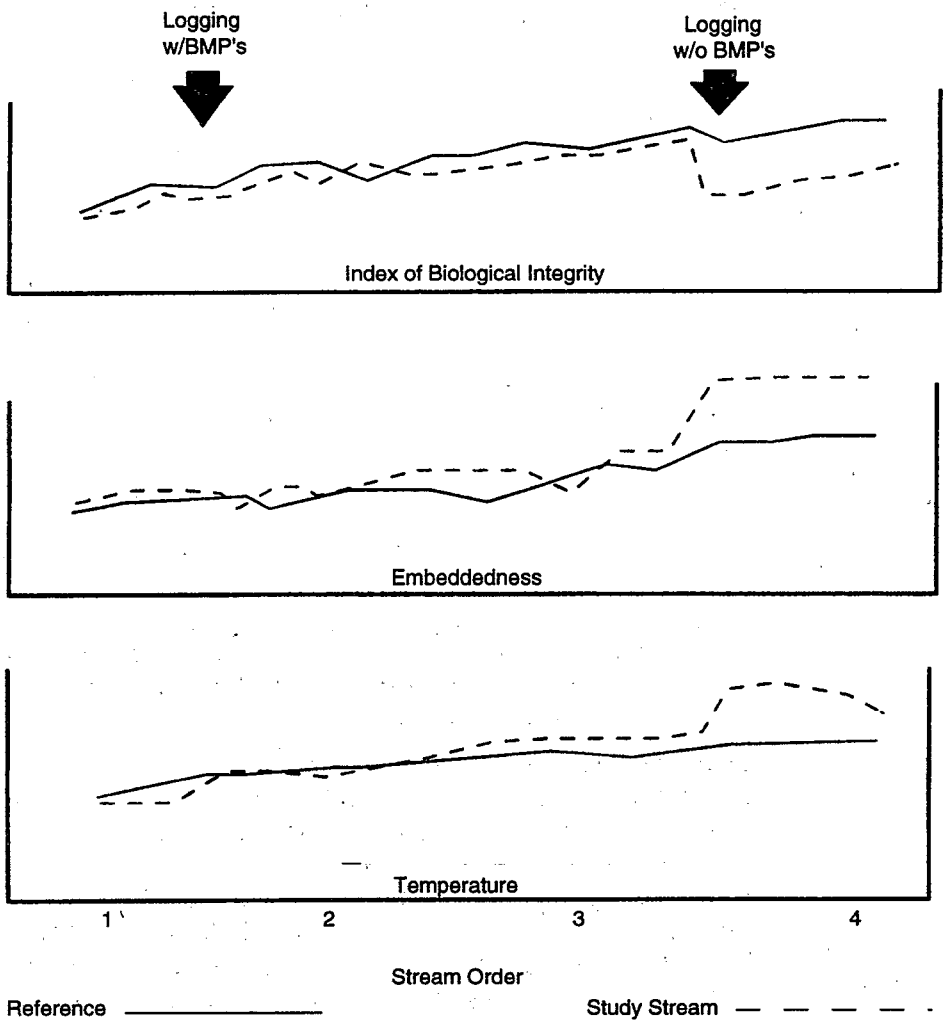

Figure 26. Monitoring effectiveness along a stream continuum.

BMP Effectiveness by Stream/Ecosystem Class

At the BMP Effectiveness III and IV levels, another approach to evaluating BMP effectiveness might be by stream ecosystem class. Some of the methods suggested in this report were tested in a forest setting on streams with the same stream/ecosystem classification. The test was conducted on the Silver Creek Research watershed in the Boise National Forest. Tim Burton, Boise National Forest in Idaho, agreed to test the method and prepare a report (Burton 1993), which is excerpted below (his citations are not included in References):

The Silver Creek study was established in the early 1960's to obtain information on the environmental impacts of logging and associated road construction. The study encompasses 7 small watersheds located in the headwaters of Silver Creek, a tributary of the Middle Fork Payette River, on the Boise National Forest (Figure 27). These third order watersheds are all similar in size, aspect, and elevation, and ideal for evaluating the effectiveness of various silvicultural management practices to protect water quality and supporting beneficial uses of water.

To date, most research in Silver Creek has focused on measures of erosion, sedimentation, and water yield and their responses to silvicultural treatments and road building. Sediment collection dams, stream gages, and precipitation gages were established in the mid 1960's. Measurements of sediment and water yield have been continuous since that time. The watersheds were calibrated for a period of 10 years prior to logging treatments, which began in 1976. Road construction was initiated in 1980 on three of the watersheds, the others remained unroaded. Three unroaded watersheds were logged by helicopter. Logging continued on various watersheds until 1985.

A wide variety of treatments, including ground and helicopter yarding, various stand opening sizes, different slash disposal methods, and ranges of road standards were applied to the various watersheds. One watershed, Eggers Creek was never entered and was retained as a control for comparison purposes.

Benthic macroinvertebrates and aquatic habitat characteristics were monitored prior to logging in 1973 and 1974. These baseline data provide a unique opportunity to assess the relative logging impacts on biological integrity. Most of the same stations visited in the 1970's were re-visited in 1992 to evaluate the aquatic impacts of the various silvicultural treatments. Benthic macroinvertebrate indicators reflect changes in the aquatic ecosystem and integrate environmental impacts that have occurred over the period of logging and application of best management practices.

Monitoring Strategy

A total of 17 monitoring stations were sampled in the fall of 1992 at approximately the same locations as those of Hart and Brusen (1976). Within the 7 study watersheds, stations were located near the mouth, in the reach upstream of each watershed's sediment collection dam. In two watersheds, stations were also located upstream of the locations of road construction, at sites not previously sampled by Hart and Brusven (1976). Also, several sites on the mainstem of Silver Creek were sampled to evaluate habitat changes downstream. Locations of all stations are displayed on the watershed map (Figure 27).

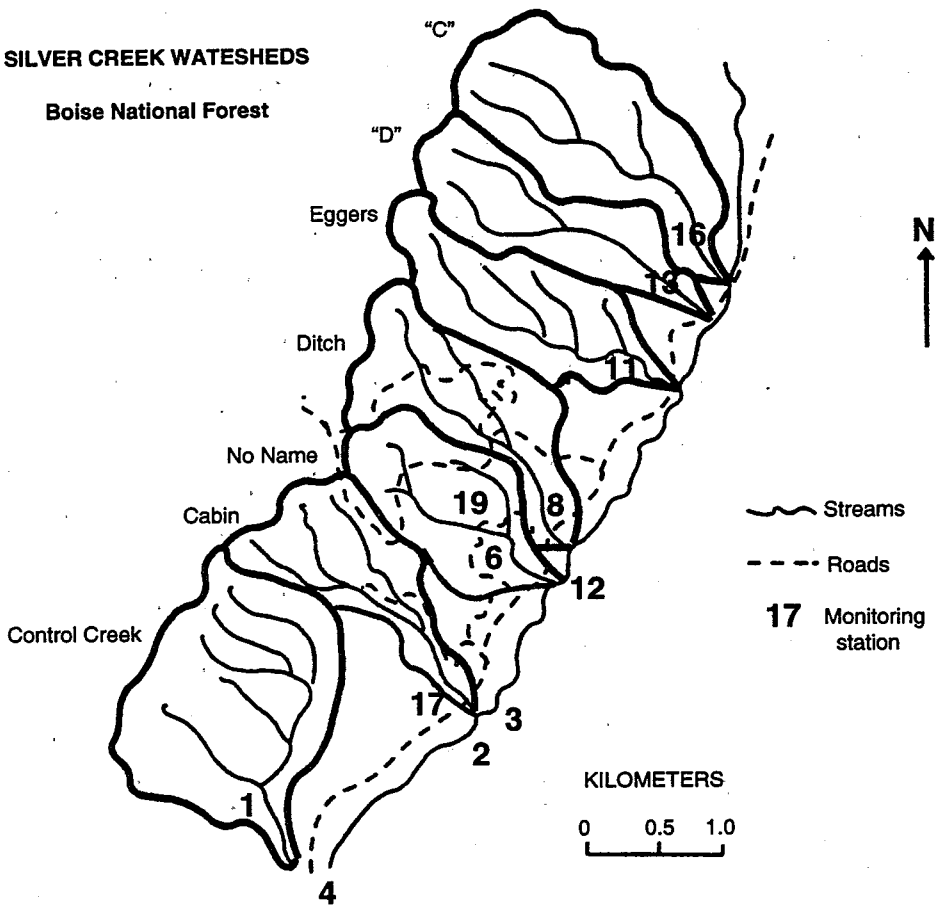

Figure 27. Map of Silver Creek watersheds.

Samples collected in the fall of 1992 on treatment watersheds are compare to the fall samples before treatment (1974), as well as those in the control watershed after treatment to assess relative effects on the biological health of the streams. Thus, three cause-and-effect strategies are used: (1) Before-after; (2) Treatment-reference; and (3) Upstream-downstream.

A variety of silvicultural BMP's have been applied to the research watersheds during the past 12 years. This project evaluates the relative effectiveness of these treatments in protecting aquatic ecosystem health. Details of the BMP's and research observations on erosion and sediment production through the early 1980's are contained in cited literature. Recent sediment production estimates are not available for the Silver Creek watersheds, but will be reported in 1994.

Description of Treatments and Their Effect on Sediment Yield
The following is a watershed-by-watershed summary of logging and roading treatments applied over the ten year period (1976-1985) and associated sediment yields.

1. ***"C" Creek Watershed.*** *This basin was logged in 1982. The harvesting method consisted of logging very small patches and yarding by helicopter. No roads were constructed. Slash was hand piled and not burned. Sediment yields were not significantly increased as a result of these treatments.*

2. ***"D" Creek Watershed.*** *This watershed was logged by clearcutting 33 percent of the area in 2 to 25 acre patches. Buffer strips of about 75 feet were left adjacent to perennial streams in the watershed. These buffer strips were left undisturbed, except for the removal of any trees expected to die before the next scheduled harvest. Any logging slash that fell in streams was removed. Yarding was accomplished by helicopter, slash was lopped and scattered and not burned, and no roads or skid trails were constructed in the basin.*

   Although the overall sediment yield in the basin was not significantly increased by these activities, a sapping (piping) failure was detected below one of the clearcut patches in this watershed (Megahan and Bohn, 1989). This type of mass failure is caused by increased soil water seepage into headwater, zero-order channels. The reduction in evapotranspiration within the clearcut increases ground water movement downslope, causing soil in adjacent swails to become saturated. Mass wasting results when large volumes of material become saturated and begin to flow downslope in liquid or slurry form below the sapping zone. A new channel is created by headward migration of the failure scrap or headcut.

   The failure in this watershed commenced in 1984 and produced 38 cubic meters of sediment in 1988. In 1986, this particular sapping failure accounted for 36 percent of the total annual sediment yield from

the watershed. Channel sediment may have increased slightly downstream after logging as a result of the mass failure.

3. ***Eggers Creek Watershed.*** *This watershed has not been entered for logging or road construction. The watershed is used as a control for comparison purposes.*

4. ***Ditch Creek Watershed.*** *Ditch Creek has a long history of development. In 1933 a low standard road of about 9 miles was constructed entirely within the 250 acre watershed. Megahan et al. (1983) estimated erosion rates on this road of 225 tons/hectare/year (the 45 year average). This is more than 1000 times greater than natural slope erosion on undisturbed forest slopes in the Silver Creek area. During the calibration period, prior to logging and roading treatments in the study, sediment yields were consistently higher here than other watersheds in study (averaging .1 cubic meters/hectare/year as compared with .01 to .07 on undisturbed watersheds).*

   A .6 mile connector road was constructed in Ditch Creek in 1980 using routine erosion control measures for that time. Routine erosion controls included: no road surfacing, uncompacted fills, no downspouts, no treatment of cutslopes, and dry seeding of fill slopes. The road was about 25 wide, had an average cutslope length of 40 feet and fill slopes of about 35 feet, and average grade of 8 percent (Megahan, et al., 1986). During construction, sediment yields were more than 5 times the average for pre-logging years. Most of the eroded sediment (77%) was stored on the slopes below the road. About 10 percent of the road sediment became channel sediment which has continued to be reflected in increased sediment yield and affect aquatic habitat since road construction.

   Small clearcut logging (1-5 acres) occurred in Ditch Creek in 1987. About 26 percent of the area was logged. Ground skidding using caterpillar tractor and rubber tire skidder was used on gentle slopes, and helicopter logging was used elsewhere. The slash was machine piled and burned.

   Because of the development history in this watershed, it has produced significant increases in sediment yield at the watershed outlet.

5. ***No Name Creek Watershed.*** *About one mile each of connector and local roads were constructed in the watershed in 1980. Routine erosion control practices, as described for Ditch Creek Watershed were used on roads in this watershed. Sediment yield increases were equally significantly after logging: about 5 times the pre-logging sediment yields.*

   No Name Watershed was selection logged in 1985. Skidding was by a combination of ground and helicopter depending on proximity to roads. Routine slash disposal practices were used.

Significant increases in sediment were produced mostly as a result of road system development.

6. ***Cabin Creek Watershed.*** *This drainage was logged and roaded using maximum erosion control efforts. About .5 miles of connector and 1.3 miles of local roads were constructed in the basin in 1980. Road erosion control included: asphalt surfacing on the connector road, rock surfacing on the local roads, mulching and revegetation of the fill slopes, energy dissipators on culverts, road location following contours (minimizing cuts and fills), rolling grade (drainage controls), and few culverts (thus little concentrated drainage from the road). Sediment production at the drainage outlet in Cabin Creek, however was significantly greater than the pre-development average, but only by one order of magnitude. This compares to 5 orders of magnitude in Ditch and No Name watersheds.*

   Cabin Creek Watershed was logged in 1987. Both ground and helicopter skidding were employed in the watershed. About 28 percent of the area was harvested in small clearcuts (1-4 acres). Slash was piled but not burned.

   This watershed has not experienced the severe sediment increases observed in several of the other developed watersheds. Maximum erosion control efforts minimized the adverse affects, even though development has been at a comparable level.

7. ***Control Creek Watershed.*** *Control Creek was clearcut in 1976. Three clearcuts totaling 100 acres of old growth ponderosa pine and Douglas-fir were helicopter yarded. This represented 23 percent of the watershed area, and all logging was on south-facing slopes, which in the batholith are sensitive to erosion and sedimentation.*

   Logging in Control Creek was not associated with any road building or construction of landings. Trees were removed from areas located at least 100 feet from perennial streams. Buffers adjacent to streams were left undisturbed, except for removal of some dying trees. Slash was lopped, scattered, and then broadcast burned in the fall of 1976 and winter of 1977. The fire severity was not great and only 50 percent of the clearcuts were actually burned.

   Significant increases in sediment production occurred on this watershed after logging. Megahan (1987) indicated that sediment yields showed a statistically significant increase of around 100% over previous levels resulting from timber harvest. Accelerated sediment production persisted at least 10 years after logging, and showed no signs of abating throughout the decade. Research showed that 6 percent of the sediment production resulted from mass erosion associated with the clearcuts, and that 94 percent of the increased sediment yield was from surface erosion caused by long-term exposure of bare soil in logged and burned clearcuts. The harsh site conditions on these south

slopes restricted vegetation regrowth allowing accelerated erosion conditions over long periods of time.

Prescribed burning on these sites negated the beneficial effects of helicopter logging. In such situations, no burning is advisable. Erosion on these fragile landscapes increased about 80 times over pre-logging erosion rates causing serious increases in sediment yield.

Summary: *Only 2 of the 7 watersheds showed no significant increases in sediment production after 1976, the control watershed Eggers Creek and "C" Creek. The latter watershed was logged, but no roads were constructed, trees were removed by selection logging, and slash was not burned. The "2" and Cabin Creek watersheds experienced relatively small increases in sediment yield. On these, intensive application of BMP's seemed to prevent severe sedimentation increases. Trees were yarded using helicopter methods mostly. Roading in Cabin Creek was associated with maximum erosion controls. The three other watersheds experienced significant sediment yield increases resulting from clearcut logging, burning of slash, tractor yarding, and normal road construction practices.*

Based on the research results, Burton rated the effectiveness of BMP application within the various watersheds as follows:

Undeveloped:	*Eggers Creek*	*Rating = 3*
Highly effective:	*"C" Creek*	*Rating = 2*
Moderately effective:	*"D" Creek* *Cabin Creek*	*Rating = 1*
Ineffective:	*Ditch Creek* *No Name Creek* *Control Creek*	*Rating = 0*

METHODS

At each of the stations shown on the map (Figure 27), biological monitoring was conducted to evaluate BMP effectiveness. Monitoring was conducted at the BMP Effectiveness III level. This project employed the basic protocols for rapid bioassessment (Plafkin et al. 1989) and uses local adaptations for metrics assessment (Clark 1992).

At each station, three replicate samples were collected using a standard Hess sampler at a series of randomly selected riffles. One hundred organisms were composited from the samples for rapid bioassessment. Identifications were made on several samples in the field and all samples were identified in greater detail at the Idaho Department of Health and Welfare laboratory in Boise. Field identification was conducted to compare metric results from lab identification and determine relative cost effectiveness of the two methods. At several sites, two composite samples were collected and transported to the lab to evaluate spatial variability and sample size effectiveness.

Biotic indices were computed for the macroinvertebrate samples at each station. These included: Total number (abundance), total taxa, total EPT taxa, % EPT, % chironomids, % sediment indicators, % clean water indicators, HSI tolerance, BCI tolerance, % scrappers, % filterers, % shredders, ratio of scrappers/filterers, and ratio of % EPT/chironomids.

Habitat evaluations were performed at each site using the standard RBP habitat assessment approach as modified by Clark (1993). Such habitat evaluations are based on qualitative ratings of cover, substrate embeddedness, canopy, channel condition, pool-riffle ratio, bank stability, and riparian vegetation condition. Because sediment is so important in the present evaluation, a simple quantitative technique was applied to estimating surface fine sediments. On each plot sampled for macroinvertebrates, a gridded plexiglass plate was placed on the water surface and each grid intersection over fine sediment was tallied to determine percent surface fines. Plots were averaged to determine site conditions.

RESULTS

Spearman ranked correlations were used to evaluate possible relationships between BMP effectiveness and habitat and biological parameters. Results are presented in Table 15. Based on these data, the effectiveness of silvicultural management practices can best be evaluated by substrate fine sediment, habitat condition (score), percent of EPT insects (stoneflies, caddisflies, and mayflies), percent shredders, percent filterers, and percent scrapers.

Among the habitat factors evaluated, percent fine sediment had the strongest correlation to BMP effectiveness. The relationship is displayed in Figure 28. Ineffective BMP's (ratings less than 2) were associated with higher sedimentation, and substrate fines greater than 25 percent.

Table 15. Spearman Rank Correlations—BMP effectiveness in the Silver Creek watersheds

PARAMETER 1	PARAMETER 2	rho
BMP effectiveness	% Fine Sediment	-.82
BMP effectiveness	Habitat score	.69
BMP effectiveness	Pool/Riffle ratio	.77
BMP effectiveness	Bank Stability	.69
BMP effectiveness	Channel alteration rating	.77
BMP effectiveness	Substrate Stability	.77
BMP effectiveness	% EPT insects	.64
BMP effectiveness	% Filterers	-.66
BMP effectiveness	% Shredders	.76
BMP effectiveness	Ratio: Scraps/Filts	.66

The relationship between EPT insects and BMP effectiveness reflects the impact of sedimentation on this taxonomic group. As sediment production increases, the numbers of these insects decline while the numbers of Diptera (especially the midges) increase. As shown in Figure 28, as substrate fines increase above 30 percent, the numbers of EPT decline to less than 55 percent of the sample. The effect on EPT relates to a shift in composition as shown in Figures 29 and 30. Figure 29 indicates that as fine sediments increase, the abundance of the stonefly, Peltoperlidae decreases. Figure 30 shows an opposite relationship to Chironomidae.

The stonefly Peltoperlidae has an affinity to forested streams with abundant shade and canopy cover as well as course substrates. It was suggested as an excellent indicator of the effect of logging practices in earlier Silver Creek studies by Brusven (1978). This insect, along with the Chironomidae (midges) are easily recognized in the field. Because of their relative abundance, they constitute major factors in computation of the EPT index.

Burton evaluated the relationship between biological factors and substrate fine sediments using the Spearman rank correlation. Results are presented in Table 16. Percent fines have a negative correlation with pool/riffle ration, bank stability, channel alteration, and the number of EPT taxa. These relations suggest that as sedimentation increases, adverse channel changes are concurrent with declines in EPT taxa.

Table 16. Spearman Rank Correlations—Percent fine sediment in the Silver Creek watersheds

% Fine Sediment	*Pool/Riffle ratio*	*-.79*
% Fine Sediment	*Bank Stability*	*-.80*
% Fine Sediment	*Channel alteration*	*-.88*
% Fine Sediment	*% EPT Taxa*	*-.60*

Percent EPT also indicated a high correlation to habitat score, bottom cover, pool/riffle ration, channel alteration, and substrate stability. No other biotic indices correlated significantly with the habitat parameters. Because EPT insects can be estimated in the field, this index has tremendous potential for evaluating BMP's in a cost-effective monitoring program.

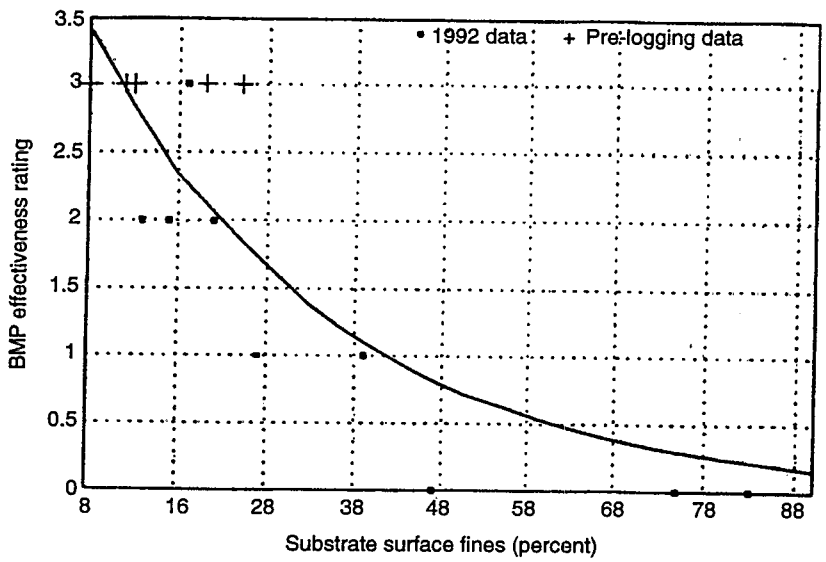

Figure 28. Relationship between percent substrate fines and BMP effectiveness rating for sediment control, Silver Creek watersheds.

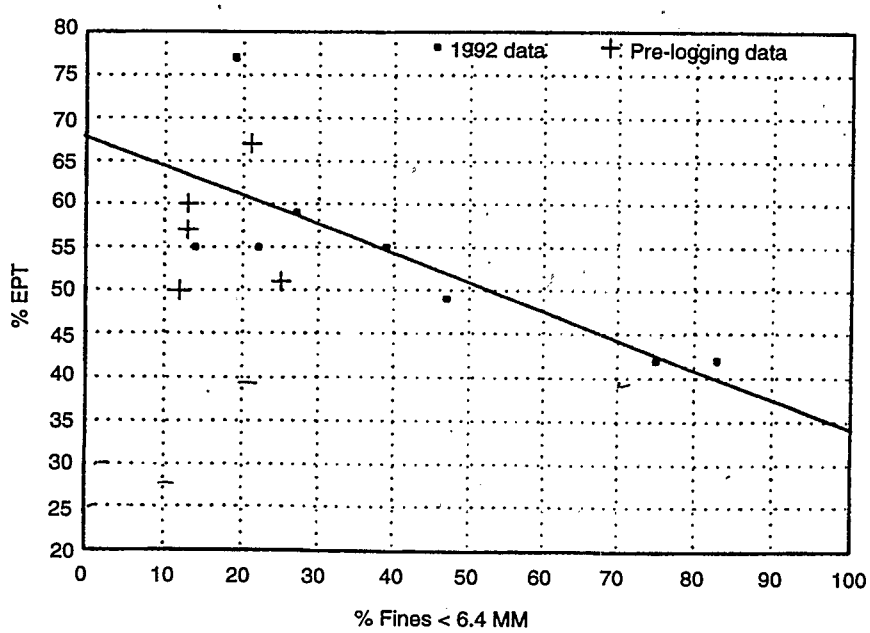

Figure 29. Relationship of percent EPT to percent fine sediment in substrate, Silver Creek watersheds.

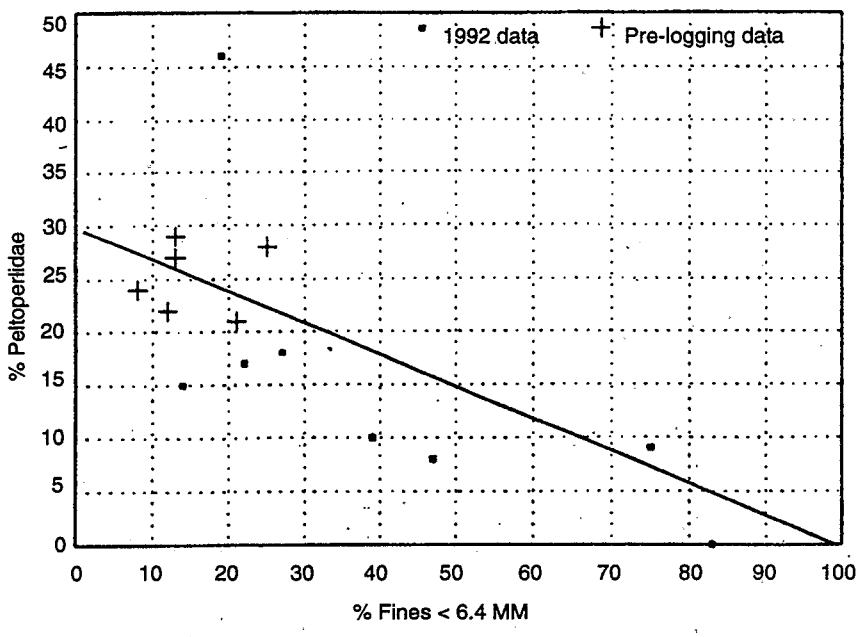

Figure 30. Relationship of percent Peltoperlidae to percent fine sediment in substrate, Silver Creek watersheds.

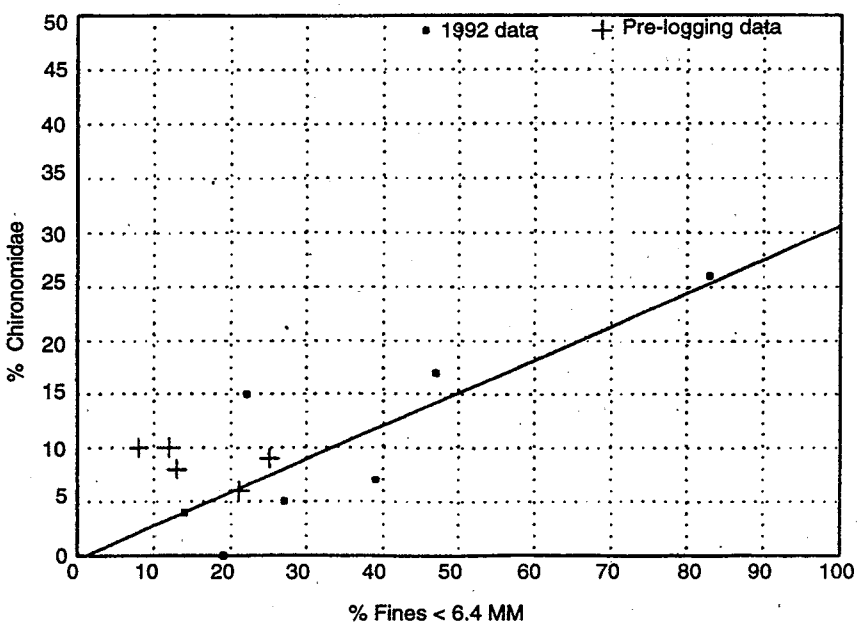

Figure 31. Relationship of percent Chironomidae (Midges) to percent fine sediment in substrate, Silver Creek watersheds.

Reference conditions for percent fines, percent EPT, percent Peltoperlidae, and percent Chironomidae can be gleaned from Burton's data in Figures 28 to 31. The prelogging data plus data from Eggers Creek (control) could represent reference site values for the four variables and the range of these scores can be used to define reference conditions. The reference condition for fine sediments is 12-25%, for percent EPT it is 50-77%, for percent Peltoperlidae it is 22-48%, and for percent Chironomidae it is 1-10%. If the study stream score for percent fine sediment and percent Chironomidae exceed the corresponding reference condition, then management activities in the watersheds impaired these variables. If the study stream scores for percent EPT and for percent Peltoperlidae fall below their reference conditions, impairment has occurred.

The effectiveness of management activities in the 6 developed watersheds can be compared to these reference conditions (Table 17). The small-patch clearcuts yarded by helicopter without roading in "C" Creek resulted in the 4 variables falling within reference conditions, suggesting that this form of logging and associated BMP's protected water quality.

Cabin Creek had well-designed and well-located access roads that incorporated sound road surfacing and stabilization BMP's. Twenty-eight percent of the watershed was clearcut and slash was not burned. Sediment yield was increased some, but the 4 water-quality scores fell within the reference conditions. Diagnosis: water quality was protected by this type of management and quality of BMP's.

Table 17. Management and BMP effectiveness evaluation of the 6 developed and the control Silver Creek watersheds (bold values indicate departure from reference conditions)

Watershed	Sediment Yield Increase	Percent Fines	Percent EPT	Percent Peltoperlidae	Percent Chironomidae
"C" Creek	None	14	51	25	5
"D" Creek	Some	**39**	55	**19**	7
Egger Creek (Control)	None	19	77	48	1
Ditch Creek	5 Times	**75**	**42**	27	2
No Name Creek	5 Times	**83**	**42**	**24**	**38**
Cabin Creek	Doubled	25	55	35	12
Control Creek	80 Times	**47**	**49**	**19**	11

"D" Creek was logged by helicopter and was unroaded. A buffer strip was left between the logged area and the stream. The land form and reduced evapotranspiration resulted in a small landslide after logging. An increase in sediment yield from the landslide resulted in the percent fine sediment and the percent Petloperlidae scores departing from reference conditions (see bold values in Table 17). These responses would suggest impairment. "C" Creek was managed essentially the same way, with good results. "D" Creek results suggest that some refinement in BMP's to minimize the potential of landslides may be in order.

In Ditch and No Name Creeks, the lack of erosion control on roads clearly increased sediment yields and resulted in increased percent fines and reduced EPT, both of which are outside reference conditions. The combination of management activities and lack of implementation of road BMP's appears to have impaired water quality significantly.

Logging a south aspect and broadcast burning in Control Creek also resulted in water-quality impairment, based upon water-quality variables.

The Silver Creek project clearly demonstrates multi-factor monitoring and how various management activities can be evaluated for effectiveness within an ecoregion and for a stream/ecosystem class.

An Example of BMP Effectiveness I Monitoring

Walsh (1992) evaluated the change in biological diversity from above to below harvest areas using the Izaak Walton League (1990) biological diversity index. Above the harvest site, the biological diversity index was 26, but below it was 5. Both samples were taken in similar riffles. Temperature, pH, conductivity, total dissolved solids, turbidity, nitrate, and ortho-phosphate concentrations were similar at both stream locations. Walch states: "The only observable impact on stream quality was a large haul road that began far up the slope and emptied directly into the small stream. No water bars, seeding, pipe culvert or means for stream crossing protection were present and heavy sediment flow had occurred in the past, evidenced by the road condition. Possibly the erosion from this road had contributed significantly to stream degradation."

Determining Effective Life of Best Management Practices

Questions often asked include: Are forestry best management practices effective for as long as it takes for a management area to recover and stabilize? How long are BMP's effective? How can we determine how long they are effective?

Several factors determine how long BMP's are effective, including: (1) quality of BMP implementation, (2) climate, (3) rate of vegetative recovery, (4) type of soils, and (5) the occurrence of large storms.

The quality of BMP implementation refers to meeting design specifications. Waterbar installation is a good example. Are they the right height? Right angle? Do they discharge onto undisturbed soil? To accurately evaluate the design life of BMP's, they must be installed according to design specifications. If not, and they fail, it is not the fault of the design.

Climate greatly influences the effective life of BMP's. Snow-melt hydrology is different from rainfall hydrology. Rainfall amounts and erosive energy vary widely from region to region. Mediterranean climates have wet winters and droughty summers, whereas in other regions precipitation occurs relatively uniformly throughout the year. Some regions have intensive thunderstorms that can produce several inches of rain in a short period. The form of precipitation, associated erosion, and surface runoff influence the life of BMP's.

Closely associated with climate is the rate of vegetative recovery. In the arid West, vegetation may take decades to invade and stabilize disturbed areas, while in the humid South it may take only a matter of months to a few years, depending upon the disturbance.

For example, the filtering capability of streamside management zones (SMZ) is related to the rate of vegetative recovery above them. SMZ's are designed to absorb surface runoff and trap eroded soil before it reaches a stream. SMZ's can only trap so much sediment before they become ineffective. The period of vegetative recovery determines how long SMZ's are subjected to accelerated sedimentation. Establishing SMZ's to specific design specifications may be effective in the South where vegetative recovery is fast, but these specifications may not be effective in the arid West where recovery is slow.

The type of soil also helps determine the effective life of BMP's. Soil stability, erodibility, compactibility, moisture-holding capacity, nutrient content, permeability, and hydrophobic conditions all influence BMP's effective life. Soil stability refers to risk of mass movement. Soil erodibility influences the rate of soil detachment and amount of sediment from a site. Soil-moisture holding capacity and nutrient content influence the rate of vegetative recovery. The susceptibility of a soil to compaction influences surface runoff and the rate of vegetative recovery. Some soils have high infiltration rates, others low, and some are hydrophobic (water repellent).

Major storms can shorten the life of BMP's. A major storm can overload filter strips with sediment, wash out drainage structures and waterbars, cause landslides, etc. With expected precipitation and storms, effective life is usually a matter of a few years. However, if a rare storm occurs during the recovery period, the BMP's may fail.

Determining Effective Life Under Normal Weather Patterns: A BMP is considered effective if, throughout its life, it meets the physical, chemical, and biological goals or standards for water. Some BMP's may be effective for the first year or so but gradually fail in subsequent years. The question is, how long is a BMP effective? The following monitoring procedure will answer this question for

normal weather patterns. Evaluating BMP effectiveness when rare, major storms occur is discussed later.

When and how long to monitor depends on response lag, that is, how long after the management activity the expected response occurs and on the expected duration of the management impact and the longevity of the BMP's. For example, monitoring sediment from surface erosion could begin with the initiation of the activity and continue for a few years, while landslides may not develop until 5 years after logging so monitoring for landslide sediment could be delayed.

The methods for determining effective life and BMP effectiveness are the same. The choice of monitoring level again depends on the issues or need; effective life can be determined at all four levels of monitoring. The procedure is as follows:

1. Use reference and study streams, and classify the stream/ecosystem to ensure streams or stream segments are as similar as possible.

2. Select precipitation gages near reference and study streams. Analyze the precipitation record to determine if the rainfall and snowstorms are representative of those expected during a normal recovery period for the management activity. If a rare storm has occurred, the study may need to be moved to area of "typical" precipitation for the normal vegetative recovery period.

3. Locate management areas of various ages adjacent to streams.

   Logged areas in the South require 3 to 4 years to revegetate and stabilize, so areas 1, 2, 3, and possibly 4 years old should be located for study. In the West, it may take 10 years to recover, so logged areas 1-10 years old could be selected for study, or areas 1, 3, 5, 7 and 9 years old could be evaluated. The management areas need to be similar in geology, slope, soils, aspect, and vegetation.

   The alternative is to implement a management activity and BMP's and monitor conditions on-slope and in-stream until the area is completely healed.

4. Using on-slope monitoring methods, evaluate the quality of BMP implementation, trace sediment to point of deposition or to the stream, and estimate or measure soil erosion, etc. Evaluating on-slope data over time reveals trends in the amount of bare soil, litter decomposition, revegetation, rill and gully formation, and sediment in filter strips, etc.

5. Monitor the physical, chemical, and biological integrity of streams to document the health of the streams associated with each management age. Compare study streams with reference streams to evaluate the amount of impact on the physical, chemical, and biological (PCB) integrity. The effective life of BMP's is time between implementation to when BMP's break down and result in unacceptable impacts to streams.

In-stream impacts may not occur until a disturbance is a few years old. Coupled with on-slope monitoring data, this information may document when BMP's break down and begin to cause unacceptable impacts.

6. If the apparent effective life is not at least as long as the recovery period, review and revise the BMP implementation standards. The on-slope information should show what, where, and how BMP's fail and suggest how to modify them to extend their effective life.

Why is extending effective life important? Water-quality agencies usually expect water-quality goals or standards to be met at nearly all times, which means they will accept only very short-term departures, meaning days or weeks, not years.

Major Storms and BMP Effectiveness

Foresters often wonder how large a storm BMP's will withstand. Some BMP's have storm size considered; for example, culverts are typically designed to handle a storm with a 25- to 100-year-return period. But what about other BMP's? How do we determine how large a storm waterbars, filter strips, road drainage, etc. can withstand? Will filter strips handle runoff and sediment from a storm expected to occur annually, every 2 years, every 5 years, every 10 years, every 25 years, every 50 years, or every 100 years? Management cannot wait 10 to 100 years to find out.

The solution to this problem is to keep track of weather events in the region or State. Usually within the region or State, a large storm will occur every few years. Using data on the precipitation amount and duration, the investigator can determine the return period for the storm from weather maps, by calculating storm-frequency distributions, or by referring to State climatical publications or Weather Bureau publication TP-40.

The area affected by the storm can be mapped. Within the affected area, forest management areas with BMP's can be located by contacting local foresters. The same monitoring logic should be used as for evaluating the effective life of BMP's: age of management areas and BMP's needs to be considered; reference and study streams need to be assessed for physical, chemical, and biological integrity. The on-slope evaluation of BMP's needs to be fairly detailed; tracing soil movement and runoff is vital because runoff, erosion, and sediment are the primary variables related to with storm size and BMP effectiveness.

On-slope monitoring will determine if waterbars survived and are functional, if runoff and sediment overwhelmed filter strips, if revegetation treatments held, if landslides occurred and in association with what practices or land features, etc.

In-stream concerns may be channel and stream-bank stability, transport of large woody debris, substrate stability, habitat changes or damage, flushing out of benthic invertebrates and fish, etc.

Invertebrate populations can be dramatically reduced with substrate movement and habitat loss from high-flow events. If a study stream has low benthic invertebrate and fish populations, reduced habitat, and impaired channel stability, is it due to BMP failure or natural processes associated with major a major storm? Assessing the reference stream may reveal similar conditions, suggesting the storm as the culprit.

Investigating areas subjected to various storm sizes and building a database of on-slope and in-stream information can help determine the size of storms individual BMP's can withstand in a region. Some BMP's may work in storms with 5- or 10-year returns, while others fail. A 25-year storm may wash out or breach waterbars, while rolling dips work; filter strips may not keep runoff and sediment from entering the stream. With a 100-year storm, all BMP's may fail.

But does the apparent on-slope failure of BMP's really matter? Reference and study streams may have essentially the same biota, habitat, and channel conditions. In such a case, management impacts are insignificant and are masked by the huge impacts from the storm.

Monitoring areas exposed to 1- to 10-year storms may reveal that proper implementation of BMP's does protect the water quality during such storms. Maybe 10- to 25-years storms cause some BMP's to fail, resulting in stream impairment. For storms of intervals greater than 25 years, all BMP's may fail, but their impacts may or may not be obscured by the natural processes.

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Appendix

The following material was taken directly from MacDonald et al. 1991. This material reviews the 29 parameters affected by forest management activities and was developed for the Pacific Northwest. It is presented without modification.

The material has been reviewed by the Task Force and the general principles apply to most situations in the rest of the country. However, the discussion of the relationship of variables to beneficial uses may vary in other regions because of differences in warmwater fisheries, wetlands, wildlife, and other uses.

The discussion of water-quality standards is general, and the user of methods in this document should use State water-quality standards to assess BMP effectiveness. Also, the States have established methods for sampling various parameters and these methods should be used whenever possible in evaluating BMP effectiveness. This will result in data comparable with State water-quality criteria.

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MONITORING GUIDELINES TO EVALUATE EFFECTS OF FORESTRY ACTIVITIES ON STREAMS IN THE PACIFIC NORTHWEST AND ALASKA

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WITH

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Monitoring guidelines.

Additional copies of this publication may be obtained from the U.S. Environmental Protection Agency, Region 10, NPS Section, WD-139, 1200 Sixth Ave., Seattle, WA 98101. Copies of the expert system may be obtained by sending a diskette formatted in MS-DOS to the same address.

PART II

REVIEW OF MONITORING PARAMETERS

1. INTRODUCTION

1.1 Purpose and Use of Part II

Part I provided guidance on the design of monitoring projects and the selection of monitoring parameters. The selection of monitoring parameters was presented as a function of the designated uses, management activities, and monitoring costs. The importance of other factors, such as geology, soils, and climate, was acknowledged, but these site-specific factors must be considered on a project-by-project basis and could not be incorporated into the tables developed in Chapter 4. An interactive, PC-based expert system, based on essentially the same selection process in Tables 2-4 in Part I (pages 39, 41 and 43, respectively), has been developed and is available from EPA's regional office in Seattle (Part I, Section 5.2).

Both the tables in Chapter 4 and the expert system result in a set of recommended parameters (Section 5.1 in Part I). The recommended parameters can be characterized as those parameters that are (1) relatively sensitive to the various management activities listed in Table 5 (pages 50-51), (2) closely related to the most common designated uses of water in the Pacific Northwest and Alaska, and (3) cost-efficient. The expert system is both more flexible and more specific than Table 5, in that the user can simultaneously select multiple management activities and designated uses, and specify the frequency of monitoring, access during high flows, and allowable costs of data collection and analysis. This additional information results in a list of recommended parameters that is more directly applicable to particular situations.

At the end of the initial selection process, the recommended monitoring parameters—either from Table 5 or the expert system—must then be evaluated individually and collectively to determine which parameters should be incorporated into the monitoring project. This "final" selection of monitoring parameters must draw upon professional judgment and consider the availability of existing data. (As discussed in Chapter 2 of Part I, the development of an effective monitoring project is an iterative process, and the initial selection may need to be modified as data and experience are accumulated.)

Often it will not be desirable or cost-effective to monitor all the parameters suggested by the tables or the expert system. Many of the 30 monitoring parameters are closely related, and in such cases an explicit choice should be made. Some of the more common pairs or groups of parameters that may overlap are suspended sediment and turbidity; nitrogen and phosphorus; channel characteristics (e.g., pool parameters and thalweg profile); and bank stability and riparian canopy opening. The potential overlap between parameters was a major rationale for preparing Tables 6a and 6b (pages 62-65). These tables qualitatively evaluate how changes in each parameter affect all the other parameters. Hence Tables 6a and 6b provide one means to both help determine the extent of the interactions between two parameters, and obtain a preliminary indication of the possible redundancy between any two parameters.

One can argue that no two parameters are completely redundant, and the inherent problems and uncertainties associated with collecting, analyzing, and interpreting field data mean that all the parameters recommended in Table 5 or by the expert system should be used. While this argument has some validity, it does not recognize the very real constraints of time and money. Clearly more data from more parameters will facilitate understanding and a more precise evaluation of changes in the stream channel and in water quality, but no monitoring project is free of cost constraints. Hence there is a need to balance idealized data needs with real-world constraints of personnel time and external costs.

The review of individual monitoring parameters that comprises Part II is a second and more comprehensive means to facilitate the selection of the most appropriate monitoring parameters. Although reading the section on each recommended parameter requires more effort on the part of the user, the additional information should lead to a

more informed and better decision.

In addition to furthering the parameter selection process, a second purpose of Part II is to summarize current knowledge on each parameter. This should help the reader understand the rationale, possibilities, and constraints for monitoring each of the 30 parameters reviewed. In order to facilitate the use of the *Guidelines* as a quick reference document, the review of each parameter is divided into seven subsections: (1) definition, (2) effects on designated uses, (3) response to management activities, (4) measurement concepts, (5) standards, (6) current uses, and (7) assessment. The last subsection for each parameter—Assessment—is a relatively brief, qualitative evaluation of the potential role of the parameter in monitoring. Hence the Assessment section can be read on its own as a summary of each parameter, and the reader then can refer back to the other subsections as needed for more information. Extensive references are provided in each of the first six subsections in order to direct the reader to key studies and more in-depth sources on any particular topic.

The parameter reviews comprising Part II do not detail field techniques and analytic procedures, as inclusion of this material was beyond the scope of the project and would greatly increase the size of the *Guidelines*. Instead, the Measurement Concepts subsection outlines some of the key considerations associated with measuring particular parameters, such as spatial and temporal variability, and the types of measurements that might be made within the more broadly defined parameters such as fish or riparian vegetation. Again the references cited will direct the reader to more detailed sources of information.

1.2 Selection and Organization of the Parameters in Part II

The 30 parameters are grouped into six categories (chapters):

1. physical and chemical constituents,
2. flow,
3. sediment,
4. channel characteristics,
5. riparian, and
6. aquatic organisms.

Each chapter includes reviews of 2 to 10 parameters that may vary widely in scope. Fish, for example, are considered within one section (i.e., as one parameter), even though there are many possible measurements which could be used in monitoring projects (e.g., species diversity, productivity, density, etc.). On the other hand, the 10 different parameters within the chapter on channel characteristics are much more narrowly defined.

There are two main reasons why parameters are included and grouped in what may appear to be an arbitrary or uneven manner. First, the *Guidelines* emphasize those monitoring parameters which are less known. It did not seem productive to duplicate the extensive literature on the more common and obvious water quality monitoring parameters, such as the chemical and physical characteristics of water. Second, the *Guidelines* focus on those parameters that appear to have considerable potential for monitoring the effects of forestry activities on streams, but which are not yet widely utilized. There is a strong and natural tendency to monitor those parameters with which one is familiar, and part of the rationale for these *Guidelines* is to take a fresh look at the entire range of monitoring parameters.

For many of the less well-known parameters, the potential for monitoring still needs to be rigorously evaluated. Often there is strong theoretical and practical justification, but relatively little experience or data to validate the use of a particular parameter for monitoring. This is the case for many of the channel characteristics, and the development of biological criteria is only now being addressed by the states. To a certain extent the differing emphasis on the various parameters reviewed in Part II reflects our attempt to anticipate future trends in water quality monitoring in forested areas. As more data are collected, modifications to the ranking and evaluation of different parameters will be necessary. Embeddedness (Section 5.6.2.) is a good example of a parameter that has undergone a rapid evolution over the past 5 years, and which is beginning to be more widely applied even though its usefulness and measurement techniques are still being debated.

Given the 1-year time frame for preparing these *Guidelines*, often it was not possible to review all the studies pertaining to a particular parameter. There also was a need to keep the individual review sections brief enough to be easily accessible but comprehensive enough to present the key elements. Inevitably there will be some dissent over the coverage or evaluation of a particular parameter, but it is our hope that such feelings will be channeled into a critical review of one's own experience and values, and that a perusal of Part II will lead to an improved understanding and formulation of water quality monitoring efforts in forested areas.

2. PHYSICAL AND CHEMICAL CONSTITUENTS

INTRODUCTION

The physical properties and chemical constituents of water traditionally have served as the primary means for monitoring and evaluating water quality. Parameters such as pH, dissolved oxygen, conductivity, alkalinity, nitrite-nitrogen, and biochemical oxygen demand are most commonly measured, and this is due both to their sensitivity to municipal and industrial pollution, and their importance in aquatic ecosystems. However, these same parameters may not be as useful in forested areas because of differences in the type of pollution, the rates of chemical and physical processes within the stream, and the designated uses of the water body.

The water column parameters included in these *Guidelines* were selected because (1) they are sensitive to forest management activities and can be related to the designated uses, or (2) they are commonly monitored in forest streams. A number of other physical properties and chemical constituents could help characterize water quality, and thereby facilitate an understanding of the aquatic system, but the focus of the *Guidelines* is on parameters useful for monitoring the effects of forestry activities on streams.

Temperature is a key parameter that can be significantly altered as a result of timber harvest immediately adjacent to the stream channel. Increases in peak summer water temperatures can directly affect coldwater fishes. Nitrogen and phosphorus are often limiting in aquatic ecosystems, and there are several means by which forest management activities—including forest fertilization—can increase nitrogen or phosphorus concentrations. Dissolved oxygen is another parameter that is critical to the health of aquatic ecosystems, but for a variety of reasons intergravel dissolved oxygen is more likely to serve as a useful parameter for monitoring the effects of forestry activities. Herbicide and pesticide concentrations generally need to be monitored when these chemicals are applied because of their potential effects on non-target organisms.

For each of these parameters, one or more surrogates can be monitored. The width of the riparian canopy opening (Section 6.1) is an important control on the amount of incoming solar radiation, and incoming solar radiation can be used to predict stream temperatures. Algal production (Section 7.2) may be related to nitrogen or phosphorus concentrations, while intergravel dissolved oxygen may be reduced by high levels of suspended sediment or fine bedload (Chapter 4). The point is that instead of directly measuring the parameter of interest, one can choose to monitor either those parameters which act as controlling factors, or those parameters which are sensitive to changes in the parameter of interest.

Conductivity and pH are included primarily because they are so often included in water quality monitoring projects. Both parameters are important indicators of the chemical and physical status of water, but they generally are much less sensitive to forest management activities than the other parameters mentioned above. They also are rarely limiting to the primary designated uses. Direct monitoring of pH and conductivity is important when other issues, such as acid precipitation, or other management activities, such as hard-rock mining, are of concern.

2.1 TEMPERATURE

Definition

Water temperature is an easily measured parameter that has considerable chemical and biological significance. It is measured on a linear scale in either degrees Fahrenheit (°F) or degrees Celsius (°C). Celsius is increasingly preferred and can be obtained easily from °F by the equation:

$$°C = 5/9\ (°F - 32)$$

Stream temperatures are the net result of a variety of energy transfer processes, including radiation inputs, evaporation, convection, conduction, and advection (Brown, 1983). Stream temperatures reflect both the seasonal change in net radiation and the daily changes in air temperature. These patterns of energy inputs and outputs are modified by stream characteristics such as the flow velocity, flow depth, and groundwater inflow. Typically peak daily temperatures occur in the late afternoon, and daily minima occur just before dawn. The seasonal pattern of stream temperatures generally is similar to the pattern of incoming solar radiation, but with a lag of 1 to 2 months (Beschta et al., 1987).

Relation to Designated Uses

Increased water temperatures are known to increase biological activity. A rough rule of thumb is that a 10°C increase in temperature will double the metabolic rate of cold-blooded organisms (Keeton, 1967). Salmonid egg and alevin development, and subsequent timing of emergence from gravel, have been shown to be closely associated with stream temperatures (Alderdice and Velsen, 1978). A rise in summertime water temperature resulting from forest harvest may increase the growth rate and productivity of many aquatic organisms (Beschta et al., 1987).

The optimal temperature range for most salmonid species is approximately 12-14°C. Lethal levels for adult salmonids will vary according to factors such as the acclimation temperature and the duration of the temperature increase, but they generally are in the range of 20-25°C. Salmonid eggs and juveniles are much more sensitive to high temperatures. Combs (1965) found the lethal limit of sockeye salmon eggs to be 13.5°C. Spawning coho and steelhead may be intolerant of temperatures above 10°C (Beschta et al., 1987).

Acute effects of high temperatures on fish have been well documented in laboratory studies, but little information is available on the long-term exposure of salmonids to sub-lethal temperatures. Similarly, the sub-lethal effects of altered thermal regimes due to forest harvest have seldom been documented for salmonid species. Recent studies by Holtby (1988) and Berman and Quinn (1990) are beginning to address these sub-lethal effects.

Stream temperature also can affect the behavior of aquatic organisms, but these behavioral effects generally are poorly understood or have been documented for only a few species. For example, at temperatures below about 5°C, juvenile salmonids tend to move into the gravel or other protected areas. This behavioral thermoregulation allows salmon and other fish to minimize body temperature fluctuations despite wide variations in stream temperatures (Coutant, 1969).

Temperature controls the rate of many chemical reactions. A general rule of thumb is that the rate of a chemical reaction proceeding at room temperature will double with a 10°C increase in temperature (Eastman, 1970). The equilibrium between ammonium and unionized ammonia, for example, is highly dependent upon temperature and can have a series of repercussions with regard to nitrogen cycling and water quality (Section 2.5.1). In contrast, the equilibrium concentration of dissolved carbon dioxide and oxygen in water is inversely proportional to water temperature (Sections 2.2 and 2.4, respectively).

Response to Management Activities

In many areas of the Pacific Northwest and Alaska, the forest cover provides substantial shade to streams and other water bodies. A reduction in the forest cover along streams can increase the incident solar radiation and hence peak summer stream temperatures. Complete removal of the forest canopy in the Pacific Northwest has been shown to increase the highest daily stream temperatures in the summer by 3-8°C, although daily summer minima are increased by only 1-2°C (Beschta et al., 1987).

These temperature increases are due almost entirely to the additional input of incoming shortwave radiation. Hence elevated stream temperatures may not return to pre-logging levels until the stream banks become revegetated and the input of shortwave radiation has been reduced to pre-logging levels (Moring 1975; Holtby, 1988). The thermal energy in streams is not easily lost through reradiation, convection, advection, and conduction. This means that increases in stream temperature generally are additive, and an alternation of shaded and unshaded reaches is not an effective strategy to minimize increased summer temperatures due to forest harvest (Beschta et al., 1987).

Removal of the forest canopy may decrease the minimum nighttime temperature in winter by allowing more radiation heat loss. In coastal areas this possible effect is likely to be minimal, but in colder locations clearing the riparian zone may cause increased incidence of anchor ice or freeze-up (Beschta et al., 1987). The largest changes in winter minima will occur in small, shallow, slow-flowing streams that do not have significant groundwater inflow.

Although the greatest effect of forest harvest is on summer maxima, smaller temperature changes in other seasons can have greater biological significance. On Carnation Creek in coastal British Columbia, for example, coho smolt numbers, size, and migration were affected more by small changes in late winter and spring temperatures than by the larger changes in summer temperatures (Hartman et al., 1987). Both this research and recent models indicate that alterations in stream temperatures can have a series of complex, interacting effects that we are only beginning to unravel for single-species systems. Holtby (1988) and Holtby et al. (1989) reported that habitat changes, like temperature elevation, can affect more than one life history stage and persist throughout the life cycle.

Measurement Concepts

Temperature can be measured by either a thermometer or an electronic sensor. Thermometers are relatively inexpensive but should be calibrated if accurate measurements (e.g., within 1°C) are required. Inexpensive thermometers may have measurement errors as large as 3°C (APHA, 1976). Electronic sensors have the advantage of allowing continuous monitoring.

To obtain average stream temperatures, measurements should be made in more turbulent reaches. Water temperatures near the bottom of pools can be 5-10°C cooler than the surface water (e.g., Bilby, 1984a). Usually thermal variations within a stream result from inflows of cool water sources, such as groundwater or intergravel water, into slow-moving reaches, pools, or backwater areas. In such cases a single surface temperature can be misleading. The daily fluctuations in stream water temperature also must be considered if instantaneous rather than continuous temperature measurements are being made.

Standards

EPA has established a general national criteria for coldwater fisheries. This states that the weekly average warm season temperature should (1) meet site-specific requirements for successful migration, spawning, egg incubation, fry rearing, and other reproductive functions of important species; (2) preserve normal species diversity or prevent appearance of nuisance organisms; and (3) not exceed a value more than one-third of the difference between the optimum and the lethal temperature for sensitive species (EPA, 1986b). Specific temperature standards to satisfy these criteria are left to the individual states.

Many aquatic organisms respond more to the magnitude of temperature variations and amount of time spent at a particular temperature than to an average value. For this reason temperature criteria should not only specify the maximum allowable increase in the weekly average, but also the maximum increase for shorter periods of time.

Current Uses

The dependence of stream temperatures on energy transfer processes suggests that changes in water temperatures due to forest harvest can be modeled and predicted. For reaches <1000 m in length, the change in maximum daily temperature can be predicted from the change in incoming direct solar radiation. The change in shading can be determined by evaluating the change in angular canopy density, and the procedure for doing this is discussed in detail by Brown (1983). This methodology also provides a basis for determining the width of a buffer strip needed to minimize changes in peak summer temperatures. Prediction of the change in stream temperatures due to partial removal of the streamside canopy is considerably more difficult.

The predictability of temperature increases due to forest harvest has recently led to the development of a model intended to be used for forest management purposes in Washington. The close relationship between mean stream and air temperatures is used as the core of a heat transfer model. Other factors, such as the riparian canopy, stream depth, and groundwater inflow, are incorporated as factors that affect heat inputs and outputs. The balance of these factors determines stream temperature and can permit the prediction of temperature patterns at the basin scale (Adams and Sullivan, 1988). Physical models that project changes in stream temperature due to management activities can then be used to evaluate the potential effects on fish, other aquatic organisms, and designated uses such as recreation.

At present, however, the width and canopy cover of buffer strips usually is fixed. Actual measurements rather than a model are used to determine whether a change in temperature has occurred. Temperature is often included in monitoring projects because it is relatively easy and inexpensive to monitor, and there is a widespread awareness of the lethal effects of high temperatures on coldwater fisheries. The scanty but increasing evidence for sublethal effects suggests that temperature monitoring should not be limited to those situations where forest harvest and other management activities are likely to result in near-lethal temperature increases.

Assessment

Measurement of summer and winter water temperatures is a useful approach to assessing the thermal suitability of a stream for fish. In contrast to most of the other parameters discussed in these *Guidelines*, temperature monitoring is relatively straightforward and inexpensive. In turbulent forest streams that are well shaded by riparian vegetation, relatively few measurements may be required because of the limited spatial and temporal variability. In pools and backwater areas, however, additional measurements may be necessary to determine whether these areas experience thermal stratification or are subject to cool-water inputs. Similarly, the timing and frequency of temperature measurements should be determined only after data have been collected on the diurnal fluctuations in temperature and the sensitivity of daily peak stream temperatures to short-term fluctuations in air temperature. Use of continuous recording devices eliminates the sampling problems caused by temporal variability.

Beschta et al. (1987) concluded that logging-related temperature increases generally have not resulted in significant mortality of resident salmonids. However, research has suggested that a variety of sub-lethal adverse effects may occur as a result of forest harvest, and this suggests that continued efforts to monitor stream temperature changes

may be desirable. The difficulty is not in monitoring these changes, but in predicting the biological effects in complex ecosystems.

Similarly, the postulated decline in nighttime winter stream temperatures due to forest harvest have not been verified. Although the magnitude of the change is likely to be relatively small in most cases, it may have important implications for stream icing in colder locations.

The additive nature of temperature increases and the likely importance of sub-lethal effects suggest that monitoring is needed when (1) the potential exists for large changes in water temperatures due to management activities, (2) water temperatures already are in the upper range of the acceptable temperatures, and (3) there is a potential for significant temperature increases due to the additive effects of numerous smaller increases. Care also needs to be taken in distinguishing temperature effects on aquatic organisms from other changes due to opening up the forest canopy such as increased light, increased nutrients, greater primary productivity, and alterations in the amount of large woody debris.

2.2 pH

Definition

pH is defined as the concentration of hydrogen ions in water in moles per liter (moles L^{-1}). Because the range of hydrogen ion concentrations in water can range over 14 orders of magnitude, pH is defined on a logarithmic scale as:

$$pH = \log 1/[H^+] = -\log [H^+]$$

where [H+] refers to the concentration of hydrogen ions in moles L^{-1}.

For practical purposes the parameter of interest is not the absolute concentration of hydrogen ions, but the chemical activity of those ions. In very dilute solutions the activity and concentration of hydrogen ions may be nearly equal, but this is less true as other ions are introduced into the sample. The common measurement techniques for pH are based on hydrogen ion activity, and do not directly measure hydrogen ion concentration.

Hydrogen ion activity varies with temperature, but it is not a simple linear relationship. At 24°C pure water has a hydrogen ion activity of 1×10^{-7} moles L^{-1}, so its pH is 7.0. Decreasing the temperature to 0°C decreases the hydrogen ion activity and increases the pH to 7.5. Increasing the temperature of pure water to 60°C increases the hydrogen ion activity and decreases the pH to 6.5 (APHA, 1980). Solutions with a higher ion hydrogen activity than pure water at 24°C have a pH <7.0 and are termed acidic. Solutions with less hydrogen ion activity have a higher pH and are called alkaline or basic.

It is important to understand that alkalinity and acidic factors refer not to the pH, but rather to the ability of a solution to neutralize acids and bases, respectively (Stumm and Morgan, 1981). In many cases both alkalinity and pH must be measured to properly evaluate changes in water chemistry due to natural events (e.g., erosion, variations in discharge) and human activities. In natural waters alkalinity is produced by anions or weak acids that are fully dissociated above a pH of 4.5. Methods for measuring alkalinity can be found in standard reference texts such as APHA (1989).

The most important buffering system in natural waters involves the dissolution of carbon dioxide (CO_2). Compared to most other atmospheric gases, carbon dioxide is relatively soluble, and in solution it combines with water to form carbonic acid (H_2CO_3). The equilibrium between carbonic acid and its component ions (H^+ and HCO_3^-) depends on the water temperature as well as the type and concentration of other ions. This carbonate system plays a critical role in water chemistry, and it is the presence of the dissociated carbonic acid that causes the equilibrium pH of pure water in contact with the atmosphere to be mildly acidic (approximately 5.7 pH units) (Hem, 1970).

pH generally shows a weak inverse relationship to discharge. At higher discharges rainfall or snowmelt is rapidly converted into runoff, and this reduces the concentration of base minerals. At low flows the incorporation of more dissolved materials tends to increase pH (e.g., Aumen et al., 1989).

Relation to Designated Uses

pH can have direct and indirect effects on stream water chemistry and the biota of aquatic ecosystems. A pH range from 5 to 9 is not directly toxic to fish, but a decline in pH from 6.5 to 5.0 resulted in a progressive reduction in salmonid egg production and hatching success (EPA, 1986b). The emergence of certain aquatic insects also declines below a pH of 6.5. From this and other data, EPA has concluded that pH should range between 6.5 and 9.0 in order to protect aquatic life (EPA, 1986b).

Indirect effects of pH on stream chemistry result from the hydrogen ion activity and the interactions between pH and a variety of other chemical equilibria. For example, at 5°C the equilibrium concentration of un-ionized ammonia can increase tenfold with a change in pH from 6.5 to 7.5 (Section 2.5.1). Similarly, the solubility of many metal compounds changes greatly with pH, and this is of critical importance in areas with high levels of heavy metals in bottom sediments. Carbonic acid in cool, CO_2-saturated streams can stimulate a wide range of weathering reactions, and this will affect the aqueous concentration of a number of dissolved ions (Reynolds and Johnson, 1972).

Response to Management Activities

Rigorous studies assessing the effects of forest management activities on pH are surprisingly scarce. The available data indicate that pH is not sensitive to most forest management activities. In two small watersheds in northwestern Oregon, for example, anion and cation concentrations exhibited virtually no change as a result of partial clearcutting and broadcast burning. Two-thirds of the total anionic load was due to the dissolution of carbon dioxide from the atmosphere (Harr and Fredriksen, 1988). In many cases the buffering capacity of the soil ensures that activities such as forest harvest, forest fertilization, and road building do not affect stream pH (e.g., Stottlemeyer, 1987).

Forest management activities can indirectly affect pH in several different ways. The introduction of large amounts of bark and other organic debris, for example, can influence pH by increasing the concentration of organic acids, increasing oxygen demand, and increasing CO_2 inputs due to respiration (Peters et al., 1976). The stimulation of primary production by increased light or nutrient loading can increase the diurnal variation in pH. Changes in the timing and volume of runoff (Section 3) can have a minor effect on pH. Erosion increases the concentration of dissolved solids and may alter pH, but conductivity (Section 2.3) and alkalinity are much more sensitive measures of this effect than pH.

Hard rock mining is the management activity most likely to substantially alter the pH of streams and lakes (Kunkle et al., 1987). The variation in mining and extraction methods makes generalization difficult, but highly acidic water is most likely to emanate from mine tailings and settling ponds. A reduction in pH exacerbates the problems associated with heavy metals by increasing their solubility and hence their mobility and rate of biologic uptake. Other types of mining, such as quarries, may alter pH if they increase the exposure of certain rock types, such as limestone, to weathering (Kunkle et al., 1987).

Measurement Concepts

pH can be measured either colorimetrically or electronically. Since colorimetric methods are subject to interference from turbidity, color, colloidal matter, oxidants, and reductants, they are suitable only for rough estimates (APHA, 1976). Usually pH is measured electronically with a pair of electrodes. One electrode is a constant-potential electrode (e.g., calomel or silver-silver chloride), and the indicating electrode usually is glass because it is relatively free from interference (APHA, 1989).

The variation of pH with temperature and carbon dioxide concentrations means that measurements should be made in the field immediately after taking the water sample. For accurate readings the pH meter must be temperature-compensated, and the sample should be thoroughly mixed between readings. The temperature of the sample also needs to be recorded, as the equilibrium concentrations of the different ions are temperature-dependent, and the pH meter cannot compensate for temperature-related shifts in the chemistry of the sample. The pH meter and electrodes must be regularly calibrated using solutions of known pH. In general, readings generally should be considered accurate only to the nearest 0.1 pH unit (APHA, 1989), and often should be assumed no more accurate than 0.5 pH units.

Accurate measurement of pH is particularly difficult in many forested areas because of the very low concentrations of dissolved solids. To obtain reliable data, the following points must be considered. First, the electrodes must be designed to function in waters with a low specific conductance. Second, pH electrodes tend to react more slowly in very dilute solutions, so a longer period of time is needed to obtain a stable reading. Third, waters with a very low concentration of dissolved solids (e.g., <50 µS) should not be stirred while readings are being taken because the stirring creates a streaming potential. Finally, the pH meter should be calibrated in standard solutions with a low concentration of dissolved solids. Calibration of the electrodes in buffer solutions of high ionic strength can lead to false readings. The known equilibrium of carbon dioxide in water means that distilled water saturated with air can be used to check or calibrate pH measurements in water with a very low ionic strength (S. McKenzie, U.S. Geological Survey, pers. comm.).

Water bodies with high algal growth can exhibit considerable variation in pH over a 24-hr period. Maximum pH values usually occur in the afternoon when photosynthetic activity consumes CO_2 and dissolved oxygen concentrations are at a maximum. Minimum pH values are observed at night when carbon dioxide is being released by algal respiration. In some cases it may be possible to use this diurnal variation in pH to estimate primary production.

Standards

EPA has set a pH range of 5.0-9.0 as the national criteria for domestic water supplies. A pH range of 6.5 to 9.0 has been established as the criteria necessary to protect freshwater aquatic life (EPA, 1986b).

Current Use

pH is included in many water quality monitoring programs because it is well recognized and perceived as easy to measure. Often, however, less data is available for pH than for other water quality constituents because it must be measured in the field immediately after taking the water sample.

Probably the most intensive program to monitor pH is the ongoing effort to assess the prevalence and effects of wet (e.g., rain, snow, and fog) and dry acid deposition. High-

elevation areas are of particular concern because they typically have thinner soils and less buffering capacity.

Regular monitoring of pH also is being conducted in conjunction with mining operations. This includes not only current operations, but also old tailings where an increase in acidity could adversely affect drinking water quality, fisheries, and the use of water for irrigation.

Assessment

The hydrogen ion activity or pH of a stream is an important water quality parameter. pH affects a wide variety of chemical reactions. pH levels above 9.0 and below 6.5 have an adverse effect on some life cycle stages of certain salmonids and aquatic macroinvertebrates. pH is of particular concern in areas contaminated with heavy metals, as a decline in pH can greatly increase their mobility.

pH generally is not sensitive to forest management activities. Hard rock mining is the management activity which is most likely to affect pH in aquatic systems. Abandoned mine tailings and certain other types of mining also can affect pH, and the intensity of monitoring will depend upon factors such as the type of rock being mined or disturbed, the designated water uses, and the amount of drainage from the mine or spoils.

Monitoring pH in forested areas may necessitate special procedures and equipment because most surface waters have very low concentrations of dissolved solids. Failure to acknowledge these special considerations can easily lead to unreliable data. Synoptic measurements can indicate the spatial variability of pH. Some of the differences between streams can be related back to physical factors such as climate and geology.

Temporal variation can occur on different scales. Diurnal variation is often due to primary production, while monthly and seasonal variation results from factors such as fractionation during snowmelt, changes in runoff processes, and changes in atmospheric deposition. The potential linkage between pH and discharge means that simultaneous flow measurements are needed for thorough data analysis.

2.3 Conductivity

Definition

Conductivity (or specific conductance) refers to the ability of a substance to conduct an electric current. The conductivity of a water sample is a function of the water temperature and the concentration of dissolved ions. Conductivity may not be directly proportional to the concentration of dissolved ions, as ion mobility, ionic charge, and ionic concentrations may affect conductivity in a non-linear manner (APHA, 1976). The relationship between conductivity and temperature also is slightly non-linear, as the dissociation constants of different ions vary with temperature. For dilute solutions, a 1°C increase in temperature increases conductivity by approximately 2% (Hem, 1970).

Conductance is the inverse of resistance and is measured in the reciprocal of ohms, or mhos. Conductivity is measured in terms of conductance per unit length, or mhos/cm. These units are too large for most natural waters, so the usual unit is μmhos/cm, where 10^6 μmhos is equal to 1 mhos. Conductivity may also be reported in millisiemens/meter, with 1 millisiemen/m equal to 0.1 μmhos/cm (APHA, 1976).

Pure water not in contact with the atmosphere has a conductivity of approximately 0.05 μmhos/cm. Normal distilled or deionized water has a conductivity of at least 1.0 μmho/cm, and this is largely due to the dissolution of carbon dioxide in water (Section 2.2). Melted snow in the western United States has a conductivity of 2 to 42 μmhos/cm (Hem, 1970). The range for potable water in the U.S. is 30 to 1500 μmhos/cm. The conductivity of streams emanating from forested areas in the Pacific Northwest almost always falls at the low end of that range (e.g., Aumen et al., 1989).

Relation to Designated Uses

Conductivity is an indication of the number of dissolved ions in the water. This makes it very useful for quickly assessing the quality of water for irrigation or water supply purposes, and for monitoring the total concentration of dissolved ions in wastewaters. Often a linear relationship can be established between conductivity and the major ionic species. Using conductivity as a surrogate for other ions can reduce the amount of laboratory work needed to characterize a sample and facilitate continuous monitoring (Hem, 1970).

Conductivity is at least as useful as total dissolved solids (TDS) for assessing the effect of diverse ions on chemical equilibria, corrosion rates, etc. In most cases TDS in milligrams per liter can be estimated by multiplying conductivity by an empirical factor. For natural waters this conversion factor ranges from 0.54 to 0.96, with most values falling between 0.55 and 0.75 (Hem, 1970).

For natural waters in the Pacific Northwest and Alaska, conductivity has no apparent effect on the designated uses of water. Conductivity is most likely to pose problems for irrigation and water supply purposes in downstream reaches subject to withdrawals of high quality water and inputs of poor quality return flows from agriculture and industry. The relative insensitivity of aquatic biota to conductivity is illustrated by the absence of an EPA-recommended criteria (EPA, 1986b).

Response to Management Activities

In the Carnation Creek study in southwestern British Columbia, conductivities increased in the sub-catchment, which was intensively logged and burned, and in the main

stem by a maximum of 90% and 50%, respectively. These increases were restricted to higher flows in the first 2 years after logging (Scrivener, 1988). In absolute terms, the increase in conductivity for high flow events from 20 to 40 µmhos/cm was well below the range of 50-120 µmhos observed during moderate and low flows. Other studies have found that forest harvest caused little or no change in the concentration of some of the major ions which contribute to conductivity, but they did not report on changes in conductivity *per se* (Brown et al., 1973; Harr and Fredriksen, 1988).

The management activity in forested areas which is most likely to affect conductivity is hard rock mining. Kunkle et al. (1987) recommend using conductivity as one of the key indicators of water quality. A substantial change in conductivity, or unusually high values, should spur more detailed analyses.

Measurement Concepts

Conductivity often is measured with temperature-compensated electrodes mounted to maintain a fixed distance between them. As with any electrodes, proper maintenance and calibration are essential for accurate measurements. APHA (1980) reports that measurements made by a trained operator should be within 1% of the true value, but tests of unknown samples resulted in a relative standard deviation of nearly 10%. Because conductivity is very sensitive to water temperature, the sample temperature should be recorded along with the conductance, or the conductance should be corrected to reflect a standard temperature such as 25°C.

Usually there is an inverse relationship between conductivity and discharge (e.g., Keller et al., 1986; Aumen et al., 1989). Water that is slowly transmitted to the stream (baseflow) has more opportunities to pick up dissolved ions through weathering and other chemical reactions. Water that is quickly transformed from precipitation to runoff (quickflow) tends to have fewer dissolved ions, thus causing a corresponding decline in conductivity at high discharges. This relationship between conductivity and discharge means that simultaneous discharge measurements are needed to properly interpret conductivity data.

Standards

No standards for conductivity have been established or proposed.

Current Uses

Conductivity is often included in water quality monitoring projects, but its use in forested areas needs to be further evaluated. In areas where conductivity between surface and groundwater differs significantly, a change in conductivity can be a sensitive indicator of groundwater seepage into stream channels. Conductivity is an excellent indicator of the total concentration of dissolved ions and thus can be a very useful indicator of mining impacts or agricultural water quality (Kunkle et al., 1987).

In forested areas the ions of primary concern, such as nitrates and dissolved plant-available phosphorus, generally are present in such low concentrations that they do not make a substantial contribution to the electrical conductivity. This makes it difficult to use conductivity as a surrogate for the more costly analyses of nitrogen and phosphorus.

Conductivity data can help characterize overall stream chemistry. Such data are particularly useful for interpreting pH measurements when both pH and conductivity are controlled by dissolved inorganic ions (e.g., in the bicarbonate-type waters that dominate in the Pacific Northwest). The relationship between pH and conductivity may be quite different in waters with high concentrations of dissolved organic matter and low concentrations of major ions (Wissmar et al., 1990).

Assessment

At the levels commonly found in forested areas, electrical conductivity alone has little or no direct effects on aquatic life. Conductivity is essentially a sum of the conductances of all the individual ionic species, so the significance of a change in conductivity depends on which ions were responsible for that change.

Forest activities can affect certain nutrients, such as nitrogen and phosphorus (Section 2.5), but these are relatively minor components of the total conductivity and generally should be measured separately. Forest activities also can affect specific conductance by altering rates of erosion and mineralization, and this proportionally increases the concentration of some of the major cations and anions (Wetzel 1975). Conductivity also can be increased by the extensive use of deicing salts or dust-reduction compounds. Although the effects of these ions on the aquatic system are believed to be negligible at the concentrations usually observed, this is one situation where conductivity monitoring may be appropriate.

The primary value of systematic conductivity measurements in forested areas is to help classify streams within a particular region or to compare streams from different regions. Such data collection efforts fall into the category of baseline monitoring and are quite distinct from monitoring to assess the effects of forest management.

Conductivity can be a useful parameter for monitoring mining impacts. It is easily measured and can serve as a surrogate for total dissolved solids or some of the major ions. If a change in conductivity is detected, more specific measurements of individual ions will be needed to determine the specific cause and predict the potential effect.

2.4 Dissolved Oxygen

Definition

Dissolved oxygen concentration refers to the amount of oxygen dissolved in water. Oxygen is a sparingly soluble gas and its concentration in water is usually measured in ppm or mg L^{-1}. The capacity of water to hold oxygen in solution (dissolved oxygen saturation) is inversely proportional to the water temperature. Increased water temperature lowers the concentration of dissolved oxygen at saturation (i.e., equilibrium with the atmosphere).[1]

The actual concentration of dissolved oxygen (DO) in water depends not only on the saturation concentration but also on oxygen sinks and sources. The primary oxygen sinks are respiration and the biochemical oxygen demand (BOD) of substances in the water. Major oxygen sources include photosynthesis and the dissolution of atmospheric oxygen in water as oxygen levels are depleted (reaeration). Higher water temperatures not only depress the concentration of dissolved oxygen in water at saturation, but also increase the rate of BOD. In general, most forest streams have cool temperatures, rapid reaeration rates, and relatively low oxygen demand; thus stream water normally is close to or at saturation. Situations in which stream water may not be near saturation include: very slow, low-gradient streams where the rate of reaeration is low; sites where fresh organic debris causes a large BOD; warm eutrophic streams where high levels of photosynthesis and respiration cause diurnal fluctuations in dissolved oxygen; and ponded sites such as those formed by beavers.

DO concentrations also can vary between the surface stream water and the water flowing through alluvial materials in the stream bed. DO within these alluvial materials is termed intergravel dissolved oxygen or intergravel DO. Oxygen replenishment to these intergravel waters comes primarily from the exchange of well-aerated surface waters with oxygen-impoverished intergravel waters. The importance of this oxygen exchange between surface and intergravel waters is a primary reason why the clogging of gravels with fines is of such concern.

Intergravel DO is controlled by the same factors as surface water, but there is no photosynthesis or reaeration. Oxygen demand comes from the fine organic debris entrained in the gravels and from the respiration of organisms living within the alluvial interstices. In spawning streams the tens or hundreds of thousands of fish eggs also can exert a measurable oxygen demand. Groundwater usually has a low concentration of DO, and areas with substantial groundwater seepage are likely to have lower concentrations of intergravel DO. For these reasons the DO concentration typically is lower within the streambed than in the adjacent stream water.

Relation to Designated Uses

DO is critical to the biological community in the stream and to the breakdown of organic material. Table 7 summarizes the biological effects of different DO concentrations in salmonid and non-salmonid waters. In salmonid streams, intergravel DO should be near saturation, or at least above minimum concentrations, to ensure normal growth and survival of eggs and alevin (Chapman and McLeod, 1987). As indicated in Table 7, high DO levels in streams and intergravel areas also are needed to sustain the more sensitive macroinvertebrates (EPA, 1986a).

Intergravel DO has been used as a surrogate for the amount of interstitial fines and as an indication of the suitability of streambed gravels for fish spawning. Note, however, that fish can greatly modify spawning site conditions, particularly the amount of interstitial fines, through the redd building process. Monitoring sites must be carefully selected to represent the actual DO concentrations that the fish eggs will experience (Chapman and McLeod, 1987).

Response to Management Activities

The Alsea watershed study in coastal Oregon indicated that heavy inputs of fine, fresh organic material, when combined with sedimentation, reduced reaeration, and increased water temperature, could severely deplete DO in small forest streams (Hall and Lantz, 1969; Wringler and Hall 1975). Subsequent research has shown that the characteristically high turbulence of forest streams rapidly replenishes DO (Ice, 1978). Current forest management techniques in the Pacific Northwest normally do not introduce large amounts of fine organic material into streams (Skaugset and Ice, 1989).

Low DO in streams is most commonly associated with major point sources such as pulp mills or municipal waste treatment facilities. Only a few examples of depressed DO related to forest management are available. In one Canadian study, for example, a stream with a slope gradient of <1% was loaded with logging debris of sufficient size and quantity to impound the stream. The fresh slash and low reaeration rate for this stream caused the DO concentration to drop to zero (Plamondon et al., 1982). Present logging practices and the increased protection for the major stream chan-

[1]The amount of oxygen that can dissolve in water increases with increasing atmospheric pressure. Dissolved oxygen saturation values (C) for different water temperatures are reported for 1 atm barometric pressure (760 mm Hg). A close approximation of actual saturation value at any pressure can be made using the equation: $C_p = C \times (p/760)$ where C_p is the saturation concentration at atmospheric pressure p (mm Hg). Generallly this pressure correction can be ignored, but it may be important for some high elevation sites. The relationship between gas solubility and pressure also may be important when effects of dam spill-ways on supersaturation disease in fish are considered.

Table 7. Biologic effects of decreasing dissolved oxygen (DO) levels on salmonids, non-salmonid fish, and aquatic invertebrates. The instream values for embryo and larval stages of salmonids were obtained by assuming that a difference of 3 mg L-1 between intergravel and instream DO would adequately maintain DO levels within the gravel (EPA, 1986a).

	Dissolved oxygen (mg L^{-1})	
	Instream	Inter-gravel
I. Salmonid waters		
A. Embryo and larval stages		
No production impairment	11	8
Slight production impairment	9	6
Moderate production impairment	8	5
Severe production impairment	7	4
Limit to avoid acute mortality	6	3
B. Other life stages		
No production impairment	8	
Slight production impairment	6	
Moderate production impairment	5	
Severe production impairment	4	
Limit to avoid acute mortality	3	
II. Non-salmonid waters		
A. Early life stages		
No production impairment	6.5	
Slight production impairment	5.5	
Moderate production impairment	5	
Severe production impairment	4.5	
Limit to avoid acute mortality	4	
B. Other life stages		
No production impairment	6	
Slight production impairment	5	
Moderate production impairment	4	
Severe production impairment	3.5	
Limit to avoid acute mortality	3	
III. Invertebrates		
No production impairment	8	
Some production impairment	5	
Limit to avoid acute mortality	4	

nels suggest that management-induced depletion of DO in stream water will occur only under unusual circumstances.

Forest management activities are more likely to affect intergravel DO through the increase in fine sediment. Everest et al. (1987) recently reviewed the linkage between fine sediment, management activities, and aquatic organisms, but provided little data on DO. An extensive review of the oxygen requirements of aquatic organisms is found in and Chapman and McLeod (1987) and EPA (1986b), but these do not relate changes in intergravel DO to management activities. Hence the cause-and-effect chain of management activities increasing fine sediment, which then decreases gravel permeability and decreases DO, must be largely inferred. Nevertheless, the linkage is sufficiently strong that Idaho has proposed intergravel DO as a sediment criteria (Harvey, 1989), and EPA has incorporated intergravel DO values into their criteria for DO (EPA, 1986a).

Measurement Concepts

Either chemical or potentiometric methods can be used to measure DO (APHA, 1989). The standard chemical method, known as the Winkler method, is based upon the oxidation of manganese, the liberation of iodine in proportion to the DO present in the sample, and then the titration of the iodine with thiosulfate. The Winkler method is very accurate provided there is no interference from suspended solids, other oxidizing agents, or certain organic compounds. Modified methods exist to reduce or eliminate each of these potential problems (APHA, 1989). The standard deviation of measurements using a standard or modified Winkler method is between 0.02 to 0.1 mg L^{-1}. Since the titration can not be performed *in situ*, it is important that the sample be collected in a manner that minimizes disturbance and gas exchange. Designs for DO sample collection devices are available (APHA, 1976).

Electrical (potentiometric) methods are based on the rate of diffusion of dissolved (molecular) oxygen across a membrane, and the resulting generation of an electrical signal. The measurement of DO by membrane electrodes is affected by both temperature and salinity, but nearly all commercially available electrodes have built-in thermistors for temperature compensation. Salinity generally is not a problem in forested areas but may need to be considered in estuaries or in streams where return-flows from irrigation result in a high concentration of dissolved solids. Provided the electrodes are properly maintained and calibrated, the potentiometric method is sufficiently accurate for nearly all field monitoring projects (accuracy of approximately ±0.1 mg L^{-1}; precision of ±0.05 mg L^{-1}) (APHA, 1989). These considerations, together with the fact that measurements can be made *in situ*, make potentiometric methods the preferred field technique.

The timing of the measurement can be important. During the day, warming of the stream water can depress the saturation concentration for DO and accelerate the rate of oxygen uptake. For slow-moving streams and rivers with high primary productivity, large diurnal fluctuations in DO concentration can result from algal photosynthesis and respiration. During the day photosynthesis in excess of respiration is a source of oxygen. At night photosynthesis ceases and respiration becomes an oxygen sink. The relative importance of the various oxygen sources and sinks must be evaluated when designing a monitoring project.

The same techniques are used to measure intergravel DO, but the collection of a representative water sample presents sample collection problems. Typically there is a great deal of spatial variability, and there is always a question as to whether one should sample from within a redd, which is most directly applicable to salmonid survival, or from a "representative" riffle, run, or pool. Moring (1975) found intergravel DO concentrations to vary from 4 to over 9 mg L^{-1} on the same day at different locations in a small, undisturbed coastal stream.

Usually intergravel water samples are obtained by placing a standpipe into the gravel some weeks or months prior to the sampling (Hoffman, 1986; Moring, 1975). Once the standpipe has been installed, a siphon can be used to remove water samples, or measurements can be made *in situ* using potentiometric methods. Skaugset (1980) used a syringe technique to rapidly extract water samples with minimal disturbance to the streambed. Two of the key principles associated with the collection of intergravel water samples are (1) minimize disturbance and gas exchange for the sample being collected, and (2) avoid disturbance to the streambed, which causes increased or decreased mixing of intergravel waters with surface waters.

Standards

Standards can either be absolute (mg L^{-1}) or expressed as a percent of saturation. Recent EPA reports discuss both the biological effects of reduced DO (EPA, 1986a) and summarize the existing state and national criteria (EPA, 1988a). The more stringent criteria are applied to those waters containing a salmonid fish population, and these state that the 1-day minimum and a 7-day mean DO concentration should be 8.0 and 9.5 mg L^{-1}, respectively. These criteria are based on the assumption that intergravel DO is about 3 mg L^{-1} less than the DO concentration in surface water, and this makes the 1-day minimum and 7-day mean intergravel DO concentration 5.0 and 6.5 mg L^{-1}, respectively. Less stringent criteria apply if only adult salmonids are present (EPA, 1986a).

State standards for DO concentrations in surface water have been established in Alaska, Idaho, Oregon, and Washington, and these vary according to the location and designated use. For basins in western Oregon with salmonids "fresh waters shall not be less than 90 percent of saturation at the seasonal low, or less than 95 percent of saturation in spawning areas during spawning, incubation, hatching, and fry stages of salmonid fish." For western Oregon basins with non-salmonids "dissolved oxygen concentration shall not be less than 6 mg L^{-1}." Some states, particularly Idaho, are considering the promulgation of a water quality criteria for intergravel DO.

Current Uses

Concern about DO is justified primarily in situations where (1) water flow is low and temperature is high; (2) the rate of energy dissipation, which accelerates reaeration, is low; and (3) oxygen sinks are high. Some examples of where this can occur are (1) slow-moving, warm streams and rivers; (2) off-channel habitat (where there is a low exchange rate for water); (3) ponded sites where water flow is slow; (4) large lakes where there is extensive log transport or storage; and (5) spawning areas and alluvial channels where the gravels are subject to high rates of sedimentation.

The conditions that cause streams to be sensitive to management impacts also make streams sensitive to natural inputs of organic material. For example, autumn leaf fall from red alder has caused a 6 mg L^{-1} oxygen deficit in a small stream in coastal Oregon (Ice, 1991).

Reductions in intergravel DO are directly related to the rate of intergravel respiration and the permeability of the gravel. Gravel permeability is sensitive to the amount of fine sediment, and this linkage has led the state of Idaho to propose intergravel DO in artificial redds as a sediment criteria in Idaho's water quality standards (Harvey, 1988). The implementation of this criteria may prove difficult because of the problems in defining key factors such as the location and size of the gravel to be used in the artificial redd. Research has shown that small changes in bed topography and gravel permeability can greatly alter the susceptibility of a redd to siltation (Cooper, 1965; Chapman and McLeod, 1987).

Assessment

Dissolved oxygen (DO) is another parameter that is easily measured and often included in monitoring efforts. While it is critical for sustaining fish and invertebrates, DO concentrations in streams are rarely a limiting factor. Forest management and harvesting activities that avoid the introduction of fresh slash into streams generally do not generate a sufficient instream oxygen demand to deplete stream oxygen concentrations. Similarly, forest practices that minimize temperature increases will help maximize absolute concentrations of DO. Conditions that contribute to a reduced concentration of DO include low flows, warm temperatures, shallow stream gradients, fresh organic matter inputs, and high respiration rates. The presence of one or more of these factors should signify a possible need to monitor DO concentrations within the water column.

Intergravel DO is more sensitive to management activities and hence potentially more useful as a monitoring technique. At least in gravel-bedded streams, any increase in the amount of fine particles is likely to adversely affect the subsurface permeability of the streambed. This reduces the rate of exchange between the intergravel and surface waters. In the absence of any change in oxygen demand in the

intergravel layer, the reduced exchange could result in a decline in intergravel DO concentrations. Numerous studies have shown that even a small decline in DO concentrations in the intergravel layer can adversely affect the reproductive success of salmonid species and the viability of other aquatic organisms. Although intergravel DO criteria have been established, very few projects have utilized intergravel DO as a monitoring parameter. This reluctance to use intergravel DO stems partly from the uncertainty regarding sample locations, and partly from the difficulty in developing acceptable sampling techniques. Nevertheless, the importance of intergravel dissolved oxygen to stream health suggests that further testing and development of this parameter is warranted.

2.5 NUTRIENTS

2.5.1 NITROGEN

Definition

Nitrogen in aquatic ecosystems can be partitioned into dissolved and particulate nitrogen. Most water quality monitoring programs focus on dissolved nitrogen, as this is much more readily available for both biological uptake and chemical transformations. Both dissolved and particulate nitrogen can be separated into inorganic and organic components. The primary inorganic forms of nitrogen are ammonium (NH_4^+), nitrate (NO_3^-), and nitrite (NO_2^-). Under certain conditions un-ionized ammonia (NH_3) also can be present.

In terrestrial ecosystems most of the soil nitrogen is associated with organic matter and is relatively immobile. Mineralization of the organic nitrogen usually converts it to ammonia (NH_3^+). Ammonium (NH_4^+) is the soluble form, and it can be taken up by plants, lost through leaching, oxidized, or fixed by exchange reactions. Normally nitrogen does not persist in the soil in the ammonia form, as it is oxidized by microbes (nitrification) first to nitrite (NO_2^-) and then to nitrate (NO_3^-). Although both nitrite and nitrate are soluble and thus subject to leaching and biological uptake, the nitrite form is relatively transient. In the undisturbed forest ecosystems of the Pacific Northwest, nearly all of the nitrate is converted into organic nitrogen by microorganisms or plants, and this completes the basic terrestrial nitrogen cycle. Losses of nitrate can occur by leaching, or by microbial reduction (denitrification) to gaseous N_2 if urea is present or conditions are anoxic (Brady, 1974; Doelle, 1975). Most of the nitrogen losses from forests to streams is in the form of nitrate (Vitousek et al., 1979), but these losses are relatively small for most undisturbed forest ecosystems (Cole, 1979; Triska et al., 1984).

In aquatic systems the inorganic forms of nitrogen (NH_4^+, NO_2^-, and NO_3^-) are subject to many of the same transformations and processes as in terrestrial ecosystems (Triska et al., 1984; Wissmar et al., 1987; Meyer et al., 1988). Nitrate is the predominant form in unpolluted waters. Un-ionized ammonia (NH_3) also may be present as an intermediate breakdown product of organic nitrogen, fertilizers, and animal wastes. Predictions of un-ionized ammonia concentrations are difficult because it is an intermediate breakdown product, and it is in a non-linear temperature- and pH-dependent equilibrium with ammonium (NH_4^+). Both ammonium and nitrate are readily taken up by aquatic biota, so an increase in nitrate concentrations tends to diminish rapidly in the downstream direction.

The riparian zone plays a critical role in nitrogen transformations as both aerobic and anaerobic conditions are present. Recent research indicates that riparian zones are important sites for denitrification (Green and Kauffman, 1989). Certain riparian plants, such as alder, can add nitrogen to the system by fixing atmospheric nitrogen, and this further complicates the interactions between the terrestrial and aquatic nitrogen cycles.

Relation to Designated Uses

Certain nitrogen compounds have toxic effects at relatively low aqueous concentrations. Nitrate has been linked to methemoglobinemia (blue-baby) syndrome in human infants at concentrations of 10 mg L^{-1} of nitrate-nitrogen (EPA, 1986b). Nitrite also will react with hemoglobin, and this can be hazardous for infants. Trout and salmon species are not as sensitive to nitrates as human infants, but nitrite-nitrogen concentrations as low as 0.5 mg L^{-1} have been shown to be toxic to rainbow trout (EPA, 1986b).

Ammonia (NH_3^-) is toxic to some aquatic invertebrates and fish at concentrations as low as 0.08 mg L^{-1}, while chronic effects occur at concentrations of only 0.002 mg L^{-1} (EPA, 1986b). The toxicity of ammonia is affected by other factors such as the concentration of dissolved oxygen, temperature, pH, salinity, and the carbon dioxide-carbonic acid equilibrium. The same factors affect aqueous NH_3 concentrations by influencing the chemical equilibrium between NH_3 and NH_4^+.

Nitrogen is one of the most important nutrients in aquatic systems. Most of the non-toxic effects of nitrogen result from the fact that increased inorganic nitrogen stimulates primary production (e.g., bacteria and algae) and possibly secondary production (e.g., macroinvertebrates and fish). However, few studies have documented an increase in primary production due to the effects of forest management on the aquatic nitrogen cycle. Studies that have attempted to analyze these more subtle effects suggest that an increase in plant-available nitrogen will increase primary productivity only if the algae are not limited by light or other nutrients such as phosphorus (Bisson, 1982). Both Lyford and

Gregory (1975) and Busch (1978) have found that nitrogen enrichment in heavily-shaded streams in the western Cascades did not enhance primary productivity. In contrast, nitrogen can be limiting in large unshaded systems like the McKenzie River in Oregon (Bothwell and Stockner, 1980).

The desirability of increased biotic production depends on the local and downstream designated uses. For many forest streams a small or moderate increase in primary production might be considered beneficial as it is likely to increase fish production. However, if plant respiration begins to deplete dissolved oxygen or results in unsightly growth of aquatic plants, this probably would be considered an adverse effect.

Increased nitrogen loading in lakes is potentially much more serious than an increase in stream nitrogen because of the potential accumulation of nutrients (Schindler et al., 1976). Over time the accumulation of relatively small nitrogen inputs may stimulate algal growth to the point where eutrophication begins and the beneficial uses such as recreation and fishing are impaired (Brown, 1988). This scenario has been documented for lakes that had a variety of nutrient inputs, but apparently not for lakes that have been subjected only to forest management activities. Studies at Lake Tahoe on the California-Nevada border, for example, indicate that logging had relatively little impact as compared to changes in land use (Goldman and Byron, 1986).

Response to Management Activities

Forest management activities can alter many parts of the nitrogen cycle, and this makes it difficult to generalize about the effects of logging, fire, erosion, and forest fertilization. Logging affects stream nitrogen by introducing organic material and sediment, and may also increase the inputs of inorganic nitrogen. In coastal British Columbia, for example, logging increased the concentration of nitrate in Carnation Creek by a factor of 2, to a maximum of 0.15 mg L^{-1} (Scrivener, 1988). Clearcutting and burning the Needle Branch catchment in coastal Oregon resulted in a fivefold increase in nitrate concentrations. However, no increase in nitrate was observed following patch cutting in the adjacent Deer Creek catchment. Maximum values in Needle Branch, Deer Creek, and the adjacent control stream were all about 3 mg L^{-1} of nitrate-nitrogen (Brown et al., 1983).

In the Bull Run watershed in Oregon, partial clearcutting caused a fourfold increase in nitrate-nitrogen when the slash was broadcast burned and a sixfold increase when the slash was allowed to decompose naturally. Maximum values followed the same pattern, with a high of 0.08 mg L^{-1} when the slash was broadcast burned and 0.27 mg L^{-1} when the slash was left to decompose (Harr and Fredriksen, 1988). In the Carnation Creek, Needle Branch, and Bull Run studies, nitrate-nitrogen concentrations returned to pre-logging levels after approximately 5 years. A more recent study has documented that, despite the relatively large increases in nitrate-nitrogen following timber harvests, the total loss of nitrogen is less than the annual input of nitrogen through precipitation (Martin and Harr, 1989).

Relatively little data is available on the indirect losses of nitrogen associated with logging. Fredriksen (1971) found that the amount of nitrogen lost in association with inorganic sediment (i.e., erosion) was larger than the amount of nitrogen lost in solution. Particulate nitrogen and dissolved organic nitrogen accounted for the majority of nitrogen lost from the experimental Fox Creek watershed following logging (Harr and Frederiksen, 1988). Generally the nitrogen associated with sediment (i.e., particulate inorganic nitrogen) is not readily available to the stream biota.

Fire also has a series of direct and indirect effects on the terrestrial nitrogen cycle (Brown et al., 1973). In general, the amount leached into the aquatic system following major fires appears to be roughly comparable to the increases due to logging (Wright, 1981). In north central Washington only traces of nitrogen were lost through leaching after a severe wildfire (Grier, 1975). A greater increase in aquatic nitrogen concentrations might be expected if substantial amounts of burned material enter the stream channel. In other cases the largest source of nitrogen following a fire could be due to increased erosion.

Plant-available nitrogen has been demonstrated to be the limiting nutrient for forest productivity in the Pacific Northwest (Gessel et al., 1979), and this has led to a number of forest fertilization programs. Most of these are on private timberland, and virtually all programs apply pelletized urea from the air at concentrations of around 200 kg/ha (Moore and Norris, 1974, cited in Norris et al., 1983). The use of pellets minimizes drift, so the delivery of fertilizer to the aquatic ecosystem occurs either from direct application, or transport by surface and subsurface runoff (Cline, 1973). Organic nitrogen in the form of urea is subject to the various nitrogen transformations, but it can also be lost as gaseous N_2 through denitrification and volatilization.

Fredriksen et al. (1975) summarized the results of several studies that monitored water quality following the application of urea. Ammonia, urea, and nitrate concentrations each peaked within a couple of days after application, although the nitrate showed a slight time lag as compared to ammonia and urea. These peaks result from the direct application of the urea pellets into the stream channels, and the relatively rapid transformation from urea to ammonia, nitrite, and nitrate. With the advent of the rainy season, a second nitrate peak was observed. The total amount of nitrogen lost to the aquatic ecosystem was estimated at 0.5% of the total applied (Fredriksen et al., 1975). More recent studies have shown that a wet season application of urea can result in a large, short-term increase in the concentration of ammonia and total nitrogen, and smaller increases in nitrate concentrations (Bisson, 1988). In more frequently fertilized watersheds, the total losses ranged up to nearly 10% of the amount applied (Bisson, 1982).

Some of the other ways in which humans increase nitrogen concentrations in streams and lakes include inadequate human waste disposal, livestock, and atmospheric fallout. Inadequate human waste disposal can result from dispersed recreation, septic tanks, and municipal wastewater treatment plants. Both dispersed recreation and septic tanks are considered nonpoint sources and require relatively intensive wet season monitoring to determine their effect on water quality. Municipal wastewater plants are point sources and therefore easier to monitor. The problem is that small rural wastewater treatment plants often cannot afford the additional treatment necessary to remove most of the nutrients, and must rely on dilution to minimize adverse effects. These considerations are taken in account when point discharge (NPDES) permits are written.

Measurement Concepts

Methods for measuring the concentration of the different nitrogen compounds in water are well known and detailed elsewhere (APHA, 1989; Stednick, 1991). An important step is to determine which nitrogen species are of most interest and to identify the measurement technique most appropriate to those species. Kjeldahl nitrogen combines both organic nitrogen and total ammonia. Total ammonia includes both ionized (NH_4^+) and un-ionized forms (NH_3). Dissolved nitrite and nitrate are often combined, as the concentration of nitrite in forested streams generally is very small. Dissolved organic nitrogen can be obtained from the difference between Kjeldahl nitrogen and total ammonia. Adding Kjeldahl nitrogen to dissolved nitrate and nitrite yields total dissolved nitrogen.

Attention also must be given to the method of expressing concentrations of the various nitrogen species. For example, a concentration of 10 mg L^{-1} of nitrate includes the weight of both the nitrogen and the oxygen atoms in the nitrate molecule, while a concentration of 10 mg L^{-1} of nitrate-nitrogen refers only to the mass of elemental nitrogen present as nitrate. The difference in the molecular weight of nitrate and nitrogen means that 10.00 mg L^{-1} of nitrate is only 2.26 mg L^{-1} of nitrate-nitrogen.

A distinction should be made between monitoring for water quality standards and monitoring to estimate total load. Monitoring for water quality standards is primarily a matter of taking samples at the times and locations where peak concentrations are expected to occur. Simultaneous discharge data is necessary for proper interpretation of the data, but not for determining whether standards are being met. Monitoring for total load requires monitoring of total dissolved and particulate nitrogen and continuous discharge measurements. As with any total load calculation, it is critical to adequately sample the high flows when the bulk of the nitrogen is being transported past the monitoring station.

Standards

The national drinking water standard for nitrate-nitrogen is 10 mg L^{-1} (EPA, 1987). A standard for nitrite-nitrogen has not been established because nitrite is such a transient form. Water bodies with high nitrite concentrations are likely to be highly polluted and not meet existing standards for other constituents such as bacterial contamination and dissolved oxygen (EPA, 1986b).

For ammonia, national criteria have been established to prevent "unacceptable" effects on freshwater organisms and their uses (EPA, 1986b). The dynamic equilibrium of ammonia with other chemical species is calculated for 1-hr and 4-day mean concentrations using formulas based on pH, temperature, and the presence or absence of salmonid species. These formulas are applicable for a temperature range of 0-30°C and a pH range of 6.5-9.0 (EPA, 1986b). Table 8 lists total ammonia concentrations that correspond to an un-ionized ammonia concentration of 0.020 mg L^{-1} for a range of common temperature and pH values (from Bisson, 1982, adapted from Thurston et al., 1974). These data indicate that the proportion of un-ionized ammonia is extremely sensitive to pH and less sensitive to temperature.

Although no national standards have been established, Cline (1973) indicated that a nitrate concentration of <0.3 mg L^{-1} would probably prevent eutrophication. In basins that have been designated as impaired, strict limitations on the total nitrogen load may be imposed (Part I, Section 1.4).

Current Uses

Many water quality monitoring programs regularly measure concentrations of one or more species of nitrogen. In undisturbed basins these data provide a baseline for comparison and an indication of long-term trends. In actively managed forested basins, forest harvest disrupts the terrestrial nitrogen cycle by increasing the amount of decomposing organic material, reducing root uptake, and changing the soil moisture regime. This can greatly increase the concentration of dissolved inorganic nitrogen—primarily nitrates and ammonium—in the stream. In many cases, however, increased leaching of nitrogen to the stream will be attenuated or completely obscured as a result of increased

Table 8. Equilibrium concentration of un-ionized ammonia in mg L^{-1} as a function of temperature and pH.

	pH		
Temperature (°C)	6.5	7.0	7.5
5	51	16	5.1
10	34	11	3.4
15	23	7.3	2.3
20	16	5.1	1.6
26	11	3.6	1.1

cation exchange reactions (Brady, 1974). Hence the principal means by which humans can increase phosphate levels in aquatic systems is by altering rates of erosion and organic matter inputs.

(Brown et al., 1973; Harr and Fredriksen, 1988). Water quality monitoring in the Carnation Creek study in southwestern British Columbia demonstrated that phosphate concentrations were unrelated to streamflow, season, or logging. Higher phosphate ion concentrations were observed after fires, but maximum concentrations of phosphate-phosphorus were still <1 µg L^{-1} (Scrivener, 1988).

uptake by aquatic plants. The complexity and interactions of the terrestrial and aquatic nitrogen cycles must be considered when attempting to relate a change in stream chemistry to a particular management activity (Meyer et al., 1988).

Monitoring of nitrogen species can be useful when undertaking a forest fertilization program. Past studies sug-

validity of the water quality model, and changes due to management actions.

Assessment

Almost any forest management activity will affect some

Part II

On the other hand, aerial application of herbicides and pesticides can have serious implications for non-target organisms. Herbicides, for example, are usually directed at broad-leaved species, and many riparian species are susceptible to, if not the target of, the commonly used herbicides. Killing the riparian vegetation can have a wide range of secondary effects, including destabilizing the stream channel, reducing the input of both fine organic material and large woody debris, and increasing stream temperatures. Similarly, the aerial spraying of pesticides can adversely affect the riparian fauna, and this can reduce the availability of terrestrial insects for fish populations.

Response to Management Activities

Herbicides, pesticides, and their intermediate breakdown products are only present as a result of human attempts to control unwanted vegetation or animal pests. Ground-based programs greatly reduce the likelihood of direct contamination of streams and lakes provided proper care is taken in the transport, mixing, application, and disposal of the herbicides and pesticides.

For aerial applications the use of spray buffer strips along the stream channels greatly reduces the exposure of the riparian zones and the stream channels. For spray buffer strips to be effective, careful consideration must be given to factors such as the droplet size, height of application, wind speed, and flight path. The requirement of spray buffer strips along fish-bearing streams usually minimizes overt damage to the riparian vegetation. In tributary channels or along unprotected streams, however, herbicide use may kill off the riparian vegetation and initiate a series of adverse effects on aquatic organisms and the stream channel.

Measurement Concepts

The critical aspect in monitoring herbicide and pesticide applications is the selection of monitoring locations and the timing of the water samples. State forest practice regulations generally have established protocols for sampling, and these represent a compromise between the need for comprehensive sampling and the costs of collection and analyses.

The usual procedure is to take one sample immediately prior to application and a series of samples at various times after application. Often an attempt is made to sample peak concentrations by estimating the average velocity in the stream and then using this to estimate when the peak concentration would occur at a sampling point 200-500 ft downstream of the spray boundary. Some states specify that samples are to be taken at specified times (e.g., 0.5, 1.0 and 2.0 hr after the cessation of spraying). To minimize the costs of analysis, some states allow a portion of each short-term sample to be combined into a composite sample. The composite sample is then analyzed, and the remaining portions of the individual samples are tested only if the composite sample exceeds some threshold.

Many states also have a procedure to qualitatively evaluate the relative risk of the application to impair water quality. The intensity of sampling is then adjusted to reflect the estimated risk. The factors used to assess this risk include the type of chemical (toxicity, persistence, and mobility), the potential drift (this is a function of the slope, slope length, irregularity of the landscape, the stream length exposed to the application, the riparian cover, droplet size, and the weather conditions), and the beneficial uses of the stream (Oregon State Department of Forestry, 1979, in Appendix A, NCASI, 1984a).

One approach to the problem of sample timing is to take a 24-hr composite sample. The 24-hr composite sample is effective as long as the subsampling interval is short enough to adequately capture higher concentrations and the analytic technique is sufficiently sensitive. Combining concentration data with discharge data allows the 24-hr mean concentration to be calculated. The 24-hr mean concentration is the basis for many of the state standards for pesticide and herbicide concentrations (EPA, 1977).

Another approach is to continuously sample the stream using a trace enrichment cartridge (NCASI, 1984b). In this method a constant flow of water is run through a cartridge designed to capture the chemical of interest. Analysis of the cartridge at the end of the sampling period, when combined with discharge data, provides a mean concentration over the entire sampling period. The technology is still being tested, but concern exists that some of the subject chemical may be slowly lost from the cartridge. Preliminary data indicate that losses from trace enrichment cartridges are a function of stream pH, the flow rate through the cartridge, and the relative concentrations of the chemical in the stream and the cartridge (G. Ice, Nat. Council for Air and Stream Improvement, Corvallis, pers. comm.).

Another approach to assess overspray and drift is through the use of spray cards or tracers. Spray cards are simply flat cards which are set out prior to aerial spraying, and visual inspection provides a qualitative indication of the amount of chemicals that reached the surface of that particular site. The short lag between application and observation means that this method can be used as a near real-time monitoring technique. Disadvantages include an increased exposure to monitoring personnel and an indication of the total input, rather than the maximum concentration, of chemicals into the stream system.

Another approach is to mix a fluorescent dye with the pesticide or herbicide and monitor dye concentrations in the stream of interest. This allows direct, real-time monitoring provided that a definable relationship exists between the chemical of interest and the dye (NCASI, 1984a). Smart and Laidlaw (1977) reviewed the use and measurement of fluorescent dyes as hydrologic tracers, and they noted that a variety of different factors can influence the measurements.

One of the few field studies to measure the aqueous concentration of both a herbicide (2,4-D) and a fluorescent dye (rhodamine WT) found that the dye peak was much sharper and higher than the herbicide peak. This difference was attributed to greater sorption of the herbicide by the organic material in the stream (NCASI, 1984a). Concerns over the mutagenic activity of rhodamine WT may limit its use as a marker for herbicides and pesticides (NCASI, 1984a).

Standards

EPA has recommended maximum allowable mean concentrations over a 24-hr period for silvicultural chemicals. These concentrations vary according to the size of the stream and the designated uses of the water body (EPA, 1977). The maximum allowable mean concentrations are based on a combination of the acute toxicity as defined by the LC-50 and a safety factor. (LC-50 refers to the concentration at which 50% of the target organisms perish within the testing period.) Recommended maxima range from one-fifth to one one-hundred thousandth of the LC-50. Most states have adopted standards based on the EPA recommendations.

Current Uses

Considerable variation exists in the intensity and type of water quality monitoring associated with the application of herbicides and pesticides. Public agencies tend to test more regularly, but they are more constrained with regard to the aerial application of forest chemicals. Testing by private industry depends upon state regulations and the perceived risk.

In most cases the procedure is to take samples at a location assumed to be well mixed and therefore representative of the entire stream cross-section. These data are useful for (1) documenting the level of chemicals in the stream system, and thereby limit future liability; and (2) evaluating the effectiveness of the application techniques in minimizing the amount of chemicals released into the aquatic system.

The relative absence of articles documenting adverse water quality effects suggests that current application procedures are effective in minimizing adverse impacts. Many of the more toxic or persistent chemicals are no longer used, and this significantly reduces the possible level of exposure. Continuing attention must be paid to minimizing drift, which can be achieved by using buffer strips and spray delivery systems that generate appropriate droplet sizes, as well as by spraying under low wind conditions (EPA, 1977). The use of buffer strips along streams and lakes is the single most effective means for minimizing both the direct and indirect adverse effects of herbicides and pesticides on water quality.

Assessment

Contamination of streams and lakes by herbicides and pesticides is unlikely except in the case of accidental spills or aerial application. Monitoring of aerially applied chemicals is sporadic, although some states have established procedures to determine if water quality monitoring is necessary. It may be questioned, however, whether the typical monitoring procedures will achieve the overt monitoring objectives.

The first objective of monitoring—to document the amount of unwanted chemicals entering the aquatic system—is probably rarely achieved because of the temporal variation in pesticide and herbicide concentrations. A few grab or pump samples may or may not capture the peak concentration. In the absence of information on the shape of the concentration curve over time, the reliability of the grab samples is very low, and minor changes in the sampling location or time could dramatically affect the observed concentration.

The second use of monitoring data is to evaluate the effectiveness of the application procedures and Best Management Practices in minimizing chemical inputs to the aquatic system. This can be done only if (1) the sampling was sufficient to determine that the concentrations did not exceed some designated level, and (2) data are available to document the conditions and methods of application. In other words, downstream data only indicate whether there was a problem. Identifying the cause of the contamination will require data on all the factors that would have affected the delivery of the herbicide or pesticide into the aquatic system. Both types of data are needed to iteratively improve the application procedures, but most agency reporting forms do not request sufficient information to carry out this kind of evaluation.

Currently available information suggests that the use of herbicides and pesticides in forested areas generally does not adversely affect the designated uses of water. The relative absence of adverse effects is at least partially due to the infrequent use of these chemicals in forested areas as compared to croplands. Most silvicultural prescriptions call for no more than one or two applications of herbicides over the entire rotation period of 60-120 years. Pesticides tend to be applied only as need requires. Although aerial applications can adversely affect aquatic and riparian ecosystems, the careful application of Best Management Practices and buffer strips should minimize the impact on most water bodies.

3. CHANGES IN FLOW

Introduction

Changes in the size of peak flows, the discharge at low flows, or annual water yield usually are not considered as water quality parameters. Nevertheless, forest harvest, road building, and other management activities can result in substantial changes in the volume and timing of runoff, and this has long been a source of public concern. Changes in the size of peak flows can have important implications for the stability of the stream channel, size and quantity of the bed material, and sediment transport rates. An increase in low flows generally will reduce peak summer temperatures and increase the available fish habitat. Changes in water yield typically are too small to be measured, but in high elevation basins with extensive hydropower development the theoretical increase in water yield can have substantial economic value. In some areas the evaluation of cumulative effects is based largely on the estimated capability of the stream channel to accommodate an increase in discharge.

Flow parameters were included in the *Guidelines* because of their potential sensitivity to forest management activities, their relationship to designated uses, and general public concern. Even if a flow parameter is not explicitly included in a monitoring project, discharge measurements are needed to interpret other data, such as turbidity and conductivity, and to calculate the total flux of nutrients, sediment, and other materials being transported by streams.

In summary, the patterns and values of discharge are important characteristics of forest streams, and they integrate all the different effects of specific management activities on the hydrologic cycle. Maintaining favorable conditions of flow was an important justification for establishing the National Forest system, and this concern persists to the present day. Forest management activities can affect discharge through a variety of individual processes, and this chapter reviews the three parameters of greatest concern.

3.1 Increases in the Size of Peak Flows

Definition

Peak flows refer to the instantaneous maximum discharge associated with individual storm or snowmelt events. The diversity of climates in EPA's Region 10 means that peak flows can result from several different types of climatic events. In the low-lying, coastal basins in the Pacific Northwest, for example, winter rainfall is the primary cause of peak flows. In many of the higher-elevation and interior areas, peak flows are generated by spring snowmelt. Other possible causes of peak flow events are summer thundershowers and rain-on-snow events. Both of these latter causes may be less common and less predictable, but in certain basins they may be responsible for the largest runoff events.

Many basins may be exposed to more than one cause of peak flows. For example, spring snowmelt may generate the peak discharge in most years for a given basin, but less common rain-on-snow events may be responsible for the largest discharge events. Prediction of the effects of forest management on the size of peak flows is complicated by the fact that forest management will have quite different effects on the size of peak flows depending upon whether the peak flows are caused by spring snowmelt, high-intensity rain storms, or rain-on-snow events. The effect of forest harvest and other management activities also will vary according to factors such as the type of yarding (tractor or cable), the density of skid trails and landings, soil type, and soil moisture content. Prediction of the effect of management on the size of peak flows therefore requires (1) knowledge of the climatological events that cause the peak flows in the basin of interest, (2) specification of the peak flows of concern (e.g., the mean annual flood or more extreme events such as the 50-year flood), and (3) specific knowledge on

how the management activities are likely to affect each of the major components of the hydrologic cycle (interception, infiltration, evapotranspiration, and snowmelt).

Relation to Designated Uses

Peak flows have important effects on stream channel morphology and bed material particle size (Chapter 5). Specifically, since higher flows move larger particles, peak flows determine the stable particle size in the bed material (Grant, 1987). Large, stable particles provide important habitat niches for invertebrates and small fish. A highly unstable bed will reduce periphyton and invertebrate production (Hynes, 1970). The size of peak flows also is important in determining the stability of large woody debris and the rate of bank erosion. Increased bank erosion and channel migration will affect the riparian vegetation and alter the amount of active sediment in the stream channel. Periods of high flow also are periods of bank building and deposition on active floodplains, especially in areas with dense riparian vegetation.

The vast majority of the sediment transport occurs during peak flows, as sediment transport capacity increases logarithmically with discharge (Ritter, 1978; Garde and Ranga Raju, 1985). The ability of the stream to transport the incoming sediment will help determine whether there is deposition or erosion within the active stream channel. The relationship between sediment load and sediment transport capacity will affect the distribution of habitat types, channel morphology, and bed material particle size (Chapter 5). Increased size of peak flows due to urbanization have been shown to cause rapid channel incision and severe decline in fish habitat quality (Booth, 1990).

A change in the size of peak flows can have important consequences for human life and property. Structures such as bridges, dams, and levees are designed according to a presumed distribution of peak flows. If the size of peak flows is increased, this could reduce the factor of safety and lead to more frequent and severe damage.

Response to Management Activities

Forest management activities can increase the size of peak flows by a variety of mechanisms, and these include the following:

1. road-building (due to both the impervious surface and the interruption of subsurface lateral flow);
2. reduction of infiltration rates and soil moisture storage capacity by compaction;
3. reduced rain and snow interception due to removal of the forest canopy;
4. higher soil moisture levels due to the reduction of evapotranspiration;
5. increased rate of snowmelt; and
6. any change in the timing of flows that results in a synchronization of previously unsynchronized flows.

By the same logic indicated in item 6 above, forest harvest may reduce the size of peak flows by desynchronizing runoff peaks (Harr, 1989). Under certain conditions forest harvest also can reduce the size of the smaller peak flows by reducing fog drip, thereby reducing the amount of soil moisture storage prior to some storm events.

Each of these mechanisms will have different effects in different seasons and in storms of different magnitudes. Sufficient care in the layout and execution of roads and timber harvest will minimize the changes in the size of peak flows from the first four runoff processes identified above. Thus in the absence of rain-on-snow events, the most dramatic changes in the size of peak flows are observed in the smaller storms in autumn or early winter, when less precipitation is needed to recharge soil moisture (e.g., Harr et al., 1975; Ziemer, 1981). Forest management activities can have a relatively negligible effect on the peak flows associated with major floods if very little of the catchment has been subjected to compaction or converted to an impervious surface.

The effects of forest management on peak flow size are quite different when the largest floods are caused by rain-on-snow events. In these areas, forest management—by increasing snowpack accumulations in openings and increasing the rate of snowmelt in clearcuts and young plantations (Berris and Harr, 1987)—can increase the size of peak flows in major flood events.

The effects of forest management activities on the size of peak flows have been studied in a number of paired watershed experiments in the Pacific Northwest and elsewhere (e.g., Harr, 1983; Bosch and Hewlett, 1982). In most cases forest harvest has been found to increase the magnitude of peak flows, and this has been attributed to soil disturbance reducing infiltration and subsurface stormflow (Cheng et al., 1975), changes in short-term snow accumulation and melt (Harr and McCorison, 1979), and soil compaction (Harr et al., 1979).

A few studies have shown no significant changes in the frequency or magnitude of peak flows (Harr, 1980; Harr et al., 1982; Wright et al., 1990). In one case the absence of an increase in the size of peak flows was due at least in part to a reduction in fog drip; one must also assume there was minimal soil compaction and soil disturbance. The lesson from these studies is that forest management can have a variety of interacting hydrologic effects, and the sum of these effects will determine whether an increase in the size of peak flows is likely (Harr et al., 1982).

Measurement Concepts

Peak flows can be identified either by continuous measurement of stage (water surface elevation) or by the use of crest stage recorders. Usually stage is converted to dis-

charge by periodically surveying the stream cross-section and measuring stream velocity at various water surface elevations. The calculated discharge is then plotted against stage to obtain a rating curve (Buchanan and Somers, 1969).

The conversion of stage to discharge is needed in order to establish a quantitative relationship between peak flows in two or more basins. Changes in the size of peak flows can then be detected by a change in this relationship. Direct comparisons of stage heights between basins is not appropriate because the relationship between stage and discharge is unique for each location and may change over time as the channel erodes, aggrades, or shifts laterally.

The comparison of discharge from similar, adjacent catchments is the most sensitive means to detect changes in the size of peak flows. Usually at least 3 years of calibration data are needed to establish a relationship capable of predicting about 70-85% of the variance in discharge. A proportionally longer calibration period will be needed to establish a valid statistical relationship for peak flows with longer recurrence intervals. The pre-disturbance discharge relationship is then used to determine if there is a statistically significant change in discharge due to management activities in one of the catchments.

An alternative to the paired-catchment approach is to relate the stage or discharge at one location to precipitation, and then assess how this rainfall-runoff relationship changes with management. The difficulty with this technique is that rainfall-runoff models are relatively crude, and the uncertainty associated with rainfall-runoff model predictions generally increases with increasing discharge. This uncertainty then makes it very difficult to identify a change in the size of peak flows due to management activities.

Direct measurement of peak flows can be obtained by continuous measurements of water level or by crest-stage recorders. Continuous measurement of discharge usually requires constructing a stilling well and establishing a stage-discharge relationship. This is relatively expensive and requires a continuing input of staff time to check on the stage recorder, establish a stage-discharge relationship, and transform the stage data to discharge.

Crest-stage recorders are much simpler, as they only record the maximum water level. In the absence of a stage-discharge relationship, the values may be difficult to interpret, as changes in channel morphology can alter the observed crest from events with identical peak discharges. Typical crest stage recorders consist of vertical tubes containing powdered cork. Small holes in the tube allow water to enter and leave the crest gages, and a ring of powdered cork is left at the highest water level occurring between observations.

A major problem in monitoring changes in the size of peak flows is the infrequent nature of high flow events. Hence sample sizes are small, and the capability to detect a statistically significant change is low. For this reason most research addressing changes in peak flows have focused on runoff events that occur several times each year. Monitoring changes in the size of peak flows associated with storms with longer recurrence intervals is much more difficult. A 5-year storm, for example, only has a 20% chance of occurring in a given year, and only a 67% chance of occurring within a specified 5-year period. Hence a very long calibration period is needed for these rarer events, and the post-harvest monitoring period is limited by the hydrologic recovery of the site to pre-harvest conditions. For this reason changes in the size of the larger peak flows generally cannot be measured directly.

Monitoring changes in the size of peak flows is also limited by the cost of establishing and maintaining stations to measure peak discharges. Continuously recording gaging stations are relatively costly. Discharge measurements during high flow events require some access to the site and a structure from which one can safely measure velocity. Crest-stage recorders are relatively simple and inexpensive, but they have a much lower sensitivity.

Standards

No standards for changes in the size of peak flows have been established or proposed.

Current Uses

The difficulties in determining a change in the size of peak flows means that this parameter is rarely included in most monitoring projects. Nevertheless, potential changes in the size of peak flows can be an important constraint to forest management (Grant, 1987), particularly in areas subject to rain-on-snow events. Hence most environmental assessments and other planning documents evaluate projected changes in the size of peak flows by extrapolating from the limited number of paired-catchment experiments that have examined the issue.

It is important to note that any change in the size of peak flows is most likely to decline in magnitude as one moves downstream. This is due to both a dispersion of the flood wave in time and the lack of change in other tributaries (i.e., a dilution effect) (Linsley et al., 1982). Proportionally larger increases in the size of peak flows will occur downstream only if the timing of peak runoff in the managed basin is altered in such a way that it becomes synchronized with peak runoff in other tributaries (Harr, 1989).

Assessment

Forest management activities can increase the size of peak flows by transforming subsurface flow to surface flow, reducing infiltration rates and soil moisture storage capacity, reducing interception losses, increasing soil moisture, and altering rates of snowmelt. The relative effects of these changes will vary by season, site, and storm size. Careful management and post-harvest rehabilitation mea-

sures can largely alleviate changes in the size of peak flows due to compaction, disruption of subsurface flow paths, and reduced infiltration rates. This means that in areas not subject to rain-on-snow events, the largest change in the size of peak flows can be limited to the first few storms following the growing season, when the higher soil moisture carryover results in a greater proportion of runoff. Major floods, such as those with a return interval of 50 years or more, should not be as greatly affected by forest management activities, as the total rainfall is normally sufficient to make up any initial differences in soil moisture content. However, if forest harvest and other management activities substantially increase the amount of compacted or impervious areas (e.g., roads, landings, and skid trails), the size of peak flows from all storms is likely to increase (Harr et al., 1979).

Forest harvest can increase the size of the largest peak flows in areas where the largest floods are caused by rain-on-snow events. This increase in the size of peak flows is due to the combination of increased snowpack (caused by a reduction in interception losses) and an increase in snowmelt due to increased turbulent heat transfer. Recent research in the Washington Cascades has indicated that harvested plots can yield up to 95% more runoff than unharvested areas, and runoff from 18- to 20-year-old plantations is around 40% higher (R.D. Harr, U.S.F.S. Pac. Northw. Res. Sta., Seattle, pers. comm.).

In summary, the effects of forest harvest on the size of peak flows is difficult to predict and measure. Providing that soil disturbance and compaction are kept to a minimum, concern over increases in the size of peak flows is appropriate primarily in areas where rain-on-snow events generate the largest flood peaks. Careful monitoring of changes in the size of peak flows could help provide some insight into the hydrologic behavior of a basin, but there are more direct and efficient ways to monitor most of the physical effects that lead to a change in peak flows.

Monitoring of changes in the size of peak flows is difficult because it requires a long-term commitment and the matching of the basin of interest to one with no land use changes or management activities. Data from past studies on small catchments indicate that monitoring the size of peak flows provides little understanding unless it is accompanied by studies documenting the probable cause(s) of any observed change. Hence, monitoring the size of peak flows is more appropriate as part of an applied research project than as a standard monitoring practice.

3.2 CHANGES IN LOW FLOWS

Definition

In most of the western U.S., the minimum streamflow is observed during the late summer and early autumn. This decline in discharge is due to a combination of low precipitation, reduced drainage from the soil and bedrock, and sustained high evapotranspiration. Removal of the forest or other vegetative cover usually results in an increase in low flows by reducing evapotranspiration (e.g., Harr et al., 1979) and secondarily, interception.

Relation to Designated Uses

Summer low flows are important primarily for maintaining aquatic habitat. An increase in low flows will increase the wetted perimeter and flow depth, and thereby provide more habitat. Increased flows will also reduce the magnitude of any temperature increase due to forest harvest, as temperature increases are highly dependent on the increase in incoming net radiation relative to total discharge (Section 2.1).

Response to Management Activities

In most small catchment studies in the Pacific Northwest forest harvest has been shown to increase summer low flows by up to 300% (Anderson, 1963; Rothacher, 1970). Although this is a large relative increase, the absolute volume of the increase is small relative to the total annual water yield (Harr et al., 1982). However, in areas where fog drip is a major hydrologic input, forest harvest can cause a decline in summer low flows (e.g., Harr, 1980). Studies in the drier, snowmelt-dominated areas of the Rocky Mountains have shown low flow increases of only 0-12% following forest harvest (Bates and Henry, 1928; Troendle, 1983; Van Haveren, 1988). The presence of a low flow increase in these more arid environments may depend on whether summer precipitation is sufficient to generate a response in streamflow.

As forest regrowth occurs the increase in low flows is diminished, and the rate at which low flows return to pre-harvest conditions can be highly variable. In coastal Oregon the harvest of a mature coniferous forest was followed by the rapid establishment of phreatophytic vegetation (red alder, cottonwoods, and willows) in and adjacent to the stream channel. Within 10 years the measured summer low flows showed no increase relative to pre-harvest conditions, and in subsequent years the summer low flows were less than predicted by the pre-harvest calibration equation. This reduction in low flows can be expected to continue until the phreatophytic vegetation is overtopped by the less water-consumptive coniferous species (Harr, 1983). Hydrologic recovery from thinning, understory removal, or burning of brush also is likely to require less than a decade.

Measurement Concepts

As was the case for peak flows, the most sensitive means for detecting a change in low flows is to establish a statistical

relationship between the discharge of adjacent catchments. A change in the relationship between the two catchments is used to demonstrate a change in low flows. The need to accurately measure relatively small discharges means the gaging stations must be carefully placed to minimize seepage, and the width-depth ratio should be as low as possible. In small streams some type of weir or flume structure is likely to be needed to obtain the necessary accuracy.

Changes in low flows generally will be more difficult to detect in larger catchments because a smaller proportion of the catchment will be harvested over a relatively short time period. Hence any increase in low flows will be subject to a dilution effect from other sub-catchments which do not have a hydrologically altered vegetation canopy.

Standards

No standards for changes in low flows have been established or proposed.

Current Uses

Monitoring stream discharge is an important component of most water quality monitoring programs. However, low flows are relatively unimportant in terms of their contribution to constituent load, sediment load, and water yield. Paired-catchment experiments have shown that 20-30% of a catchment must be cleared to obtain a measurable increase in water yield (Bosch and Hewlett, 1982). Since most long-term gaging stations are on larger catchments that do not experience such heavy harvest levels over a relatively short time period, changes in low flows are unlikely to be observed at existing gaging stations.

Little attention has been paid to monitoring changes in low flows because there is very little scope for management. Removal of the riparian vegetation usually is not a viable option because of concerns over wildlife and fisheries habitats, sediment and nutrient inputs, bank erosion, and stream temperatures (Section 6.2). Forest harvest is known to decrease evapotranspiration, and some of this water will be expressed as an increase in streamflow, but we have very limited control over the amount and timing of this increase. Although this increase in low flows may be significant in terms of increased habitat area—particularly in small streams—on larger streams the increase generally is too small to be measured. For these reasons most monitoring projects do not explicitly attempt to document any change in low flows.

Assessment

Forest harvest can cause a substantial increase in summer low flows, and this will provide additional habitat for stream biota. Increased low flows also may reduce the susceptibility of the stream to adverse temperature changes resulting from removal of the riparian canopy. Thus changes in low flows may be beneficial and of interest to managers, but low flows generally cannot be used as an indicator of water quality. To date, water rights courts have not addressed the allocation of any increase in water yield due to forest harvest. The absence of any institutional mechanism to capture the economic benefits of increased low flows, and the difficulty of measuring small increases on large basins, indicates that low flow monitoring is rarely appropriate.

3.3 Water Yield

Definition

A change in water yield represents the sum of all the individual changes in runoff over a water year. Most paired-watershed experiments have focused on changes in the total annual water yield, so there is much more data on changes in water yield than on changes in low flows or the size of peak flows.

Relation to Designated Uses

The importance of an increase in water yield depends on the timing of the increase, the uses of the water, and the extent to which the increase can be captured by storage facilities. In rain-dominated or warm snow environments, the largest relative increases in water yield usually occur during the summer and first autumn storms (Harr, 1983). The largest absolute increases occur during the fall-winter rainy season (Harr et al., 1982).

In colder, snow-dominated environments most of the increase in water yield will occur early in the spring snowmelt period because less snowmelt is needed to recharge soil moisture (e.g., Troendle and King, 1985). If there is sufficient precipitation during the summer and fall to generate substantial amounts of streamflow and maintain high levels of soil moisture, water yield increases also may be detected in these periods (e.g., Swanson and Hillman, 1977).

The significance of an increase in low flows was discussed in Section 3.2; the likelihood and significance of increasing peak flows was discussed in Section 3.1. Other than the possible increase in the size of the larger peak flows due to rain-on-snow events, the increase in fall and winter discharge from forest activities is likely to have little biological or physical significance. However, any increase in flow may be beneficial if it can be captured in a downstream reservoir and used for generating electricity, irrigation, or water supply purposes.

Response to Management Activities

Bosch and Hewlett (1982) summarized the results of 94 paired watershed experiments worldwide and found that (1) in areas with over 450 mm of annual precipitation, clearing at least 20% of the forest cover resulted in a water yield increase; (2) the increase in water yield was proportional to the average annual precipitation; and (3) the increase in water yield was quite variable but was larger in wet years, particularly in dry areas. The magnitude of the observed increases ranged from 0-660 mm per year.

In general the increase in water yield due to forest management activities will be too small to be measured. U.S. Geological Survey gaging records are regarded as excellent if they are accurate to within 5%, and most local discharge measurements will be less accurate. The imprecise nature of discharge measurements, particularly at high flows, and the fact that measurable increases occur only when at least 20% of the forest cover has been removed (Bosch and Hewlett, 1982), suggest that increases in water yield can be reliably detected only when a large proportion of the forest cover has been harvested over a relatively short time period. As one moves downstream these individual increases will be smoothed out over time and increasingly diluted (MacDonald, 1989). This is why the sustainable average increase in annual water yield in western Washington and Oregon has been estimated at ≤3-6% of the unaugmented flows, while the maximum annual increase in water yield from small clearcut catchments has ranged up to 600 mm/yr per unit area (Harr, 1983).

Measurement Concepts

By definition, water yield increases must be determined by continuous stream gaging and conversion of the observed stage to discharge. A paired-watershed approach is essential to remove the effects of climatic variability and obtain the necessary sensitivity.

Since water yield increases will tend to be lost in the downstream direction, stream gaging should be conducted as high in the watershed and as close to the management activity as possible. Accurate measurements are essential, and measurements must be made during high flow periods when the bulk of the runoff is occurring. The logistical difficulties of accurately measuring streamflow during high runoff periods in remote sites cannot be overstated.

Standards

No standards for changes in water yield have been established or proposed.

Current Uses

A change in water yield integrates all the changes that have occurred over the designated time period (usually one water year). As such, it provides little information on the physical processes causing the observed change in water yield and hence little information useful to land managers. Changes in water yield may be important for water supply purposes, but the absolute amount in larger streams is very small due to the dispersed nature of forest management.

Assessment

In general, changes in water yield are detectable only in the immediate proximity of the harvested units. Measurement errors, the lack of a perfect relationship between paired basins, downstream dilution, and the small change in total volume all preclude the detection of a change in water yield in moderate-to-large streams (e.g., larger than second or third order). This, plus the extensive information already available, suggests that monitoring of water yield is not necessary under most circumstances.

On the other hand, continuous discharge measurements may be needed to calculate the total load of critical nutrients, or as part of a project to monitor turbidity or suspended sediment. If these data are being compared with an adjacent unmanaged basin, it then may be possible to utilize these discharge data to estimate the change in water yield. However, discharge and constituent data usually are collected at only a few sites in order to estimate the total load from different sub-basins, or they are being collected upstream and downstream of a particular project. Rarely are comparable data available from an undisturbed watershed. These limitations in the statistical design of most monitoring projects, together with the absence of an unmanaged control and the difficulties in accurately measuring discharge, preclude a rigorous estimation of the change in water yield due to forest management activities.

4. SEDIMENT

INTRODUCTION

An increased sediment load is often the most important adverse effect of forest management activities on streams. Large increases in the amount of sediment delivered to the stream channel can greatly impair, or even eliminate, fish and aquatic invertebrate habitat, and alter the structure and width of the streambanks and adjacent riparian zone.

The physical effects of increased sediment load can be equally far-reaching. Fine sediment can impair the use of water for municipal or agricultural purposes. The amount of sediment can affect channel shape, sinuosity, and the relative balance between pools and riffles. Changes in the sediment load also will affect the bed material size, and this in turn can alter both the quantity and the quality of the habitat for fish and benthic invertebrates.

Many nutrients and other chemical constituents are sorbed onto fine particles, so sediment loads are often directly related to the load of these constituents. Indirect effects of increased sediment loads may include increased stream temperatures and decreased intergravel dissolved oxygen (DO).

These wide-ranging effects suggest that there are an equally broad range of techniques that can be used to assess the quantity and impact of the sediment load in a particular stream. Direct measurements include suspended sediment concentration, turbidity, and bedload. Indirect methods include measurements of channel characteristics such as the width-depth ratio, residual pool depth, bed material particle size, or the width of the riparian canopy opening (Sections, 5.2, 5.3, 5.6, and 6.1, respectively). This chapter discusses only the parameters of suspended sediment, turbidity, and bedload.

4.1 SUSPENDED SEDIMENT

Definition

Suspended sediment refers to that portion of the sediment load suspended in the water column. This, at least conceptually, is distinct from bedload, which is defined as material rolling along the bed. The relative size of particles transported as bedload and suspended sediment will vary with the flow characteristics (e.g., velocity, bed forms, turbulence, gradient) and the characteristics of the material being transported (e.g., density, shape). For the Pacific Northwest and Alaska, particles <0.1 mm in diameter (clays, silts, and very fine sands) are typically transported as suspended sediment, while particles >1 mm in diameter (coarse sand and larger) typically are transported as bedload (Everest et al., 1987). Particles between 0.1 and 1 mm are usually transported as bedload, but can be transported as suspended load during turbulent, high flow events (Sullivan et al., 1987). The process of saltation, in which particles bounce from the bed up into the water column, blurs the distinction between these two terms. Local hydraulic conditions also can cause shifts in the relative proportion and size classes of bedload and suspended sediment.

Suspended sediment also should be distinguished from wash load. The latter term refers to particles that are washed into the stream during runoff events, and that are finer than the particles found in the stream bed (Ritter, 1978). By definition the wash load is finer than the bed material load, and the wash load is considered to remain suspended the length of the fluvial system (Linsley et al., 1982). Normally the wash load is defined as particles smaller than 0.062 mm (silts and clays). The concept of wash load is rarely used by fluvial geomorphologists or fish biologists, and it is difficult to apply in the type of monitoring studies addressed in these *Guidelines*.

Relation to Designated Uses

Numerous laboratory studies have documented the adverse impacts of fine sediment on benthic invertebrates as well as salmonid reproduction and growth (Chapman and McLeod, 1987). Hynes (1970) characterizes streams with sandy beds as having the lowest species diversity and aquatic productivity. As noted in Section 2.4, fine sediments tend to fill the interstices between coarser particles, and this reduces the habitat space for small fish, invertebrates, and other organisms. An intrusion of fine particles into the bed material also reduces the permeability of the bed material, and this often results in a decline in the concentration of intergravel DO (Section 2.4). Certain invertebrate species are very sensitive to even small declines in DO, and the EPA standards for DO within the water column are set in part because of the sensitivity of invertebrates and salmonid reproduction to the concentration of intergravel DO (EPA, 1986b).

Reduced gravel permeability can inhibit salmonid reproduction by reducing the concentration of DO and by entrapping alevins or fry. In a laboratory study a substrate containing 20% fines was found to reduce emergence success by 30-40% (Phillips et al., 1975). Although other field observations support the basic link between fine sediment and a decline in salmonid reproduction, direct extrapolation of laboratory studies to the field is difficult because (1) changes in suspended sediment typically are accompanied by changes in other environmental factors; (2) different species have varying sensitivity to sediment at different life stages and under different environmental conditions; and (3) changes in behavior may help alleviate the adverse effects of increased sediment (Everest et al., 1987). These same constraints apply to studies relating the concentration of fine sediment to the growth and survival of salmonid juveniles and adults.

An excess of fine sediment can adversely affect habitat availability. The case study of the South Fork of the Salmon River (Box 3, page 17) provides one example, and similar observations have been made on other streams (e.g., Grant, 1986; Cederholm and Reid, 1987; Sullivan et al., 1987). Often, however, pool infilling is due to sand-sized particles which are considered fines by fisheries biologists, but may not be transported as suspended sediment. Thus an increase in the concentration of suspended sediment may not necessarily be correlated with a decreasing bed material particle size.

Direct effects of suspended sediment on salmonids occur only at relatively high concentrations. For example, Noggle (1978) found that the ability of coho salmon fingerlings to capture prey was reduced at suspended sediment concentrations of 300-400 mg L^{-1}. Mortality of salmonids occurs only at concentrations greater than 20,000 mg L^{-1} (Everest et al., 1987).

An increase in suspended sediment concentration will reduce the penetration of light, and a sustained high concentration of suspended sediment could reduce primary production if other factors are not limiting (Gregory et al., 1987; Section 7.2). The effect of suspended sediment on water temperature has not been well documented. EPA's *Quality Criteria for Water* notes that suspended materials will increase heat absorption, particularly in the surface layer, and inhibit mixing between the warmer surface layer and the cooler underlying waters (EPA, 1986b). Others believe that the additional heating due to suspended sediment is negligible because turbid waters have a higher reflectance. The reduced penetration of solar energy caused by an increase in suspended sediment concentration could reduce the solar heating of the bed material, but the attenuation of light energy in water is so rapid that any difference in heating would occur only in areas where the water is less than about 10 cm deep. The practical implications of an increased suspended sediment load on stream temperatures and mixing are limited by the fact that (1) most forest streams are very well mixed, and (2) suspended sediment concentrations typically are very low in summer, which is when high water temperatures are of most concern.

The concentration of suspended sediment also can affect the morphology of alluvial channels. Schumm (1972) classified alluvial streams by the proportion of bedload to suspended load. Streams with 97% or more of the total sediment load as suspended sediment had width-depth ratios <10, and sinuosities >2. In such channels an increase in the suspended load would tend, at least initially, to narrow the channel as the fine sediment is deposited along the banks. Flume studies have shown that an increase in suspended sediment concentrations causes an increase in velocity and a steeper channel gradient (Chang, 1988). An increase in fine sediment may also delay the initiation of bedload transport (Beschta and Jackson, 1979). In general, however, the concentration of suspended sediment has little influence in shaping stream channels (Everest et al., 1987).

Suspended sediment can adversely affect several other designated uses of water. High concentrations of suspended sediment can damage turbines in hydroelectric plants. Suspended matter reduces the value of water for esthetic purposes. For example, it is unacceptable in municipal water supplies for esthetic reasons; moreover, it reduces the efficacy of normal treatment procedures (EPA, 1986b).

Suspended sediment will settle out in still or slow-moving waters, and this can result in clogged irrigation canals and reduced reservoir storage capacity. In some cases, however, the deposition of suspended sediment can be regarded as beneficial. For example, deposition during high flow events provides additional nutrients and soil materials. This regular deposition is a major reason why alluvial valleys often are among the most productive and fertile farmlands.

Effects of Management Activities

Forest management activities can affect the amount of suspended sediment in streams by altering both the erosion rate and the rate of transport into the stream channel. The range of management activities, and the number of erosion and transport processes, have resulted in an extensive literature on the relationship between forest management and sediment yield. However, recent changes in forest management practices may make it impossible to directly extrapolate from previous studies, even if they were conducted in a comparable environment (Everest et al., 1987). The following paragraphs provide a brief summary rather than a comprehensive overview.

Most comprehensive studies of the effects of forest management have found road-building and road maintenance to be a primary source of sediment (e.g., Brown and Krygier, 1971; Megahan and Kidd, 1972). This sediment can be eroded from the road surface (e.g., Reid and Dunne, 1984), from road fills (e.g., Megahan, 1978), or from slope failures associated with road construction and drainage (e.g., Duncan et al., 1987; Megahan and Bohn, 1989). In most cases there is a sharp increase in sediment yield associated with road-building activities, and a rapid decline as roads stabilize (e.g., Beschta, 1978). Increased sediment yields tend to be more persistent if the erosion stems from slope failures or surface runoff associated with continued heavy traffic.

Forest harvest can increase sediment yields by a variety of processes: surface erosion from landings, skid trails, and other compacted areas; slope failures triggered by removal of the tree cover; and surface erosion from burned areas or areas disturbed by site preparation activities (Swanson et al., 1987). Surface erosion can include both fluvial detachment and transport as well as dry ravel and surface creep (Swanson et al., 1987). Historic practices of disturbing the stream channel and removing large woody debris also have been shown to increase the amount of fine sediment in the stream channel (Bilby, 1981; Megahan, 1982). Removal of, or a reduction in, the riparian vegetation is a another mechanism by which forest management activities can increase the amount of fine sediments (e.g., Platts, 1981). Grazing often exacerbates the effect of reducing the vegetative cover by simultaneously trampling the vegetation, compacting the soil, and trampling the streambanks (Gifford, 1981).

In some cases management activities may have no statistically significant effect on suspended sediment concentrations. Some of the key factors controlling the actual increase in suspended sediment are as follows: (1) the intensity of disturbance, (2) the areal extent of disturbance, (3) the proximity of the disturbance to the channel system, and (4) the storm events experienced during the periods when the site is most sensitive to erosion and mass movements (Everest et al., 1987; Swanson et al., 1987). The high natural variability of suspended sediment often makes it difficult to detect a statistically significant increase in suspended sediment from well-planned and properly executed forest harvest operations.

Measurement Concepts

Suspended sediment concentrations are determined by obtaining a water sample, drying or filtering the sample, and then weighing the residual sediment. Concentrations are typically expressed in milligrams per liter (mg L^{-1}), and this usually is equivalent to parts per million (ppm) because 1 L of water has a mass of approximately 1 million milligrams. As sediment concentrations increase, however, the density of water exceeds 1000 g L^{-1}, and this causes an increasing divergence between milligrams per liter and parts per million.

The primary problem with measuring suspended sediment is how to sample in time and space. Estimates of the total amount of suspended sediment over time often are based on a presumed relationship between the concentration of suspended sediment and stream discharge, but this is by no means constant or reliable (e.g., Ferguson, 1986). For example, suspended sediment concentrations for a specified storm event typically are much higher after a dry period than after an earlier, but recent, storm. Often suspended sediment concentrations are higher during periods of increasing discharge (i.e., the rising limb of the hydrograph) and lower during periods of decreasing discharge (i.e., the falling limb of the hydrograph). However, detailed studies indicate that this is not always the case (e.g., Rieger and Olive, 1986; Williams, 1989a). Walling and Webb (1982) discuss how the physical processes of sediment production and yield need to be taken into account to better predict sediment yield and thereby reduce the apparent variability of suspended sediment concentrations.

Suspended sediment concentrations can show considerable spatial variability. The increase in suspended sediment concentration with depth is well known (e.g., Guy, 1970), but the size and concentration of suspended sediment also can vary according to local turbulence and velocity. Thomas (1985) provides a detailed discussion of the concepts and methods of measuring suspended sediment in small mountain streams.

The concentration of suspended sediment also is highly sensitive to the method of sampling. Any sampler disrupts the flow lines, and this can bias the sample. Orifice size, length of the intake nozzle relative to the sampler, and the percent of the sample bottle filled all can influence the accuracy of the sample. The hydraulic requirements of suspended sediment samplers generally preclude sampling within 10 cm or so of the stream bottom (Guy and Norman, 1970), and this limits the accuracy of any attempt to obtain an absolute estimate of suspended sediment flux.

Suspended sediment samplers can be separated into two basic types—point-integrated and depth-integrated. Point-

integrated samplers take one sample from a particular depth, whereas depth-integrated samplers allow one to sample continuously as they are raised and lowered (Guy and Norman, 1970). Since sediment flux or sediment load is of interest in most monitoring programs, a depth-integrating sampler is preferred. New, open-frame samplers allow the use of larger, wide-mouthed plastic bottles instead of the traditional pint milk bottles.

For logistical reasons most monitoring programs are using automated pump samplers. The primary limitation of these is that the intake nozzle cannot be positioned so it will sample correctly under all conditions; thus each pump sampler is measuring something different (Thomas, 1985). For estimates of suspended sediment transport, or for correct comparisons between stations, data from the pump sampler must be adjusted according to a site-specific relationship between a depth-integrated sampler and the pump sampler (Thomas, 1985).

Calculating the sediment load or sediment flux requires continuous discharge measurements. Porterfield (1972) provides detailed information on the procedures to obtain fluvial sediment discharge data, and provides a series of plots illustrating the variation in suspended sediment concentration over individual runoff events. Recent work by Cohn et al. (1989) and Walling and Webb (1982) illustrates the difficulties of accurately predicting suspended sediment concentrations from discharge data.

Most sampling schemes take individual or composite samples at regular time intervals (e.g., daily). Since high flow events are relatively rare, a sampling system based on equal time intervals will result in a large number of samples at relatively low flows, when suspended sediment concentrations are low, and very few samples at high flows, which is when most of the suspended sediment transport takes place. This is both inefficient and results in a high level of uncertainty with regard to the total sediment load. A stage-activated system can greatly increase sampling efficiency by sampling only the higher flows.

Thomas (1985) suggests linking a microprocessor to a stream gage recorder and an automated sediment sampler in order to sample on a volume basis. This increases the number of high flow samples and reduces the number of low-flow samples, with a significant improvement in both efficiency and accuracy. While such systems illustrate the potential for improved sampling procedures, they may be too costly for most monitoring applications.

Standards

Water quality standards usually are set in turbidity units rather than the concentration of suspended sediment. The general criteria established by EPA is that "settleable and suspended solids should not reduce the depth of the compensation point for photosynthetic activity by more than 10 percent from the seasonally established norm for aquatic life" (EPA, 1986b).

Current Uses

The importance and intuitive appeal of suspended sediment make it one of the more commonly used parameters for water quality monitoring. However, in most cases discharge also must be measured at the same time. In general, sampling should also focus on the high discharge events when the majority of suspended sediment is being transported. The unpredictable and short-term nature of most high runoff events suggests that if an automatic sampler is being used to take samples at constant time intervals, it may be best to take relatively frequent samples, and then discard those that do not correspond to any runoff event. Continuous discharge data are needed to interpret the suspended sediment data and estimate sediment loads and fluxes.

Simultaneous discharge measurements may not be necessary if the monitoring objectives are relatively limited. For example, construction of a bridge during summer baseflow periods may be monitored by comparing upstream and downstream suspended sediment concentrations. Such measurements will provide some indication of the effects of the management activity on suspended sediment concentrations, but in the absence of discharge data there will be no data on the total amount of sediment released by the project, or how the total load might compare to the total suspended sediment load during different storm events.

As discussed in Chapter 3 of Part I, the rigorous assessment of management impacts on suspended sediment requires data from replicated treated and untreated sites. Ideally data collected over time are used to determine the changes due to management, while data from matched sites are necessary to account for changes in the frequency and intensity of runoff events during the monitoring period. The tremendous temporal variability in suspended sediment concentrations suggests that paired (i.e., treated and untreated) sites are necessary to detect even relatively large changes. This is the approach taken in paired-catchment studies (e.g., Brown and Krygier, 1971), but the statistical conclusions from paired-catchment studies usually are limited by the lack of replication of treated and control sites (Part I, Section 3.2). Other studies have limited their ability to detect change by simply monitoring suspended sediment at one location over time (e.g., Tassone, 1988).

If sufficient suspended sediment is available, it may be helpful to occasionally conduct particle-size analyses to more accurately understand the implications for the aquatic ecosystem. At a typical density of 2.65 g/cm^3, 1 mg L^{-1} of suspended sediment can represent 90 particles of very fine sand, 90,000 particles of medium silt, or 90,000,000 particles of fine clay. In each case, the suspended sediment concentration is identical, but the relative effects on turbid-

ity, gravel permeability, and bed material particle size will be very different.

Assessment

Suspended sediment is a very useful indicator of active erosion in a particular basin. However, the multiple processes involved in sediment storage and delivery preclude the use of suspended sediment concentrations as a quantitative measure of specific hillslope and channel processes. On the other hand, suspended sediment concentrations are very sensitive to landscape disturbance, and its conceptual simplicity gives it broad appeal.

The primary problem with using suspended sediment as a monitoring tool is its inherent variability. Representative samples are difficult to obtain, and suspended sediment concentrations vary tremendously over time and space. Thus it is often difficult to determine if there has been a significant increase in suspended sediment, and whether an observed increase is due to management activities or natural causes. These problems are exacerbated as one moves farther downstream because the impact of individual management activities is diluted and the amount of suspended sediment from other sources becomes larger.

Suspended sediment can and should be included in a monitoring plan provided it is recognized *a priori* that (1) identifying an increase in suspended sediment due to forest management requires several years of background data from the basin or site where management will occur and a similar set of data from comparable, unmanaged site(s); and (2) calculating suspended sediment fluxes and loads results in an inherent uncertainty of at least 25-50%.

Suspended sediment also is just one component of the overall sediment budget. Changes in bedload generally have the greatest geomorphic impact (Section 4.3), but these may or may not be correlated with suspended sediment (Williams, 1989b). Turbidity (Section 4.2) is highly correlated with suspended sediment, but this relationship must be determined for each basin and usually each site. As indicated above, the adverse impact of suspended sediment also is a function of the size distribution of the suspended particles.

4.2 Turbidity

Definition

Turbidity refers to the amount of light that is scattered or absorbed by a fluid (APHA, 1980). Hence turbidity is an optical property of the fluid (Hach, 1972), and an increasing turbidity is visually described as an increase in cloudiness. Turbidity in streams is usually due to the presence of suspended particles of silt and clay, but other materials such as finely divided organic matter, colored organic compounds, plankton, and microorganisms can contribute to the turbidity value of a particular water sample. Since relative proportion, size, weight, and refractive properties of these materials varies considerably, a correlation of turbidity with the weight concentration of suspended matter cannot be assumed (APHA, 1980).

Prior to about 1970 turbidity was measured primarily in Jackson turbidity units (JTU). Jackson turbidity units are determined by slowly increasing the depth of water in a clear cylinder until a candle flame placed under the bottom of the cylinder disappears into a uniform glow (Hach, 1972). Several problems are associated with JTUs: (1) usable range is 25 JTUs and greater; (2) turbidity due to dark-colored particles cannot be measured as too much light is absorbed; and (3) very fine particles are not measured (APHA, 1980). These problems have led to the widespread replacement of Jackson's candle turbidimeter with photoelectric turbidimeters.

Photoelectric turbidimeters measure turbidity in nephelometric turbidity units (NTU); they are able to accurately measure much lower levels of turbidity, and measurements generally are not affected by particle color (Hach, 1972). These properties make photoelectric turbidimeters and NTU units the preferred method for measuring turbidity in streams. The differences in measurement techniques mean that there is no standard conversion between Jackson turbidity units and nephelometric turbidity units (APHA, 1980).

Relation to Designated Uses

Turbidity is an important parameter of drinking water for both aesthetic and practical reasons. A strong public reaction can be expected to a turbid water supply, even if the water technically is safe to drink. However, suspended matter provides areas where microorganisms may not come into contact with chlorine disinfectants, so high turbidity levels may limit the efficacy of normal treatment procedures (EPA, 1986b). Small rural communities may not be able to afford the additional treatment costs necessitated by an increase in the turbidity of their basic water supply (Harvey, 1989).

Turbidity also has a direct detrimental effect on the recreational and aesthetic use of water. The more turbid the water, the less desirable it becomes for swimming and other water contact sports (EPA, 1986b). In many forested areas tourism and recreation are important components of the local economy, and increased turbidity could adversely affect the attractiveness of a water body for fishing, boating, swimming, or other water-related activities.

Most of the biological effects of turbidity are due to the reduced penetration of light in turbid waters. Less light penetration decreases primary productivity, with periphyton and attached algae being most severely affected. Declines in primary productivity can adversely affect the

productivity of higher trophic levels (Section 7.2; Gregory et al., 1987).

High turbidity levels adversely affect the feeding and growth of salmonids and other fish species. A recent review concluded that the ability of salmonids to find and capture food is impaired at turbidities in the range of 25-70 NTU (Lloyd et al., 1987). Other studies indicate that growth is reduced and gill tissue is damaged after 5-10 days of exposure to water with a turbidity of 25 NTU (Sigler, 1980; Sigler et al., 1984). At 50 NTU some species of salmonids are displaced (Sigler, 1980; Harvey, 1989).

As in the case of suspended sediment, the relationship between turbidity and water temperature is not well known. The increased absorption may or may not be balanced by an increased reflectance. EPA's *Quality Criteria for Water* (EPA, 1986b) indicates that an increase in turbidity can lead to an increase in surface water temperature and a resultant decline in the rate of mixing (NAS, 1974). Reduced mixing could trigger a series of adverse effects due to the lower concentration of dissolved oxygen in the unmixed deeper portions of rivers and lakes (Section 2.4; EPA, 1986b). Although this effect is unlikely to occur in the turbulent streams characteristic of most of the Pacific Northwest and Alaska, the increased tendency towards stratification in turbid waters could be significant in reservoirs, lakes, and other downstream areas. Higher turbidity levels also could reduce the solar heating of the streambed materials, but the high absorbtion of solar radiation in water means that this is applicable only in waters less than about 10 cm deep.

Effects of Management Activities

Most studies of the effects of management activities on streams have measured suspended sediment rather than turbidity, as suspended sediment concentrations are not dependent upon the types of materials in suspension. Suspended sediment also has the advantage of being in units that can be converted to total flux over time and then related to other components of the sediment budget (e.g., erosion processes, inchannel sediment storage, and bedload transport). Hence the effects of management activities on turbidity generally have to be inferred from the relatively numerous studies that have monitored suspended sediment concentrations. Extrapolation from these studies is usually possible because of the relationship between the concentration of suspended sediment and turbidity.

In general, the same activities that generate large amounts of suspended sediment will more or less proportionally increase turbidity. However, in watersheds with coarse soils (i.e., little clay or silt), erosion and sediment yield rates can be relatively high while turbidity levels show only a moderate increase. Conversely, watersheds which primarily have clay or clay-like sediment sources could have consistently high turbidity levels but only moderate concentrations of suspended sediment; this is reportedly the case for some of the basalt watersheds in Idaho (J. Skille, Idaho Dept. of Health and Welfare, Coeur d'Alene, pers. comm.).

One of the few studies that used both turbidity and suspended sediment to evaluate the effects of road reconstruction and timber harvest was conducted on the east side of the Cascades in Washington (Fowler et al., 1988). Road reconstruction during the summer of 1979 increased turbidity levels (in NTUs) by a factor of 25 and suspended sediment concentrations by a factor of nearly 50. During the following summer suspended sediment concentrations were elevated by about 50% as compared to the upstream control site, while there was less than a 15% increase in turbidity. In the third post-treatment year, both suspended sediment and turbidity concentrations were lower at the downstream site than at the upstream control site. Timber harvest activities using a longspan skyline system and variable-width riparian zones had no detectable effect on suspended sediment or turbidity (Fowler et al., 1988). These results suggest that, at least for the above watershed (which was described as having sandy to loamy soils), suspended sediment concentrations appeared to be more sensitive to disturbance than turbidity.

Measurement Concepts

Turbidity measurements are subject to the same considerations as measurements of suspended sediment (Brown, 1983) because the most common cause of turbidity in forest streams is suspended sediment. With turbidity, however, there is an additional source of variation due to the different substances that can cause an increase in turbidity. At a particular site, for example, high turbidity levels might be due largely to organic acids at one point in time, while at another time the turbidity might be due primarily to silts and clays from earthflows or bank erosion. This variation in the sources of turbidity complicates comparisons between sites.

Typically there is a strong relationship between turbidity and discharge. As in the case of suspended sediment, this relationship will vary by site, within storms (i.e., whether discharge is increasing or decreasing), and between storms.

The relative ease of measuring turbidity as compared to suspended sediment has led to a number of studies seeking to predict suspended sediment from turbidity (e.g., Kunkle and Comer, 1971; Beschta and Jackson, 1980). These indicate that the relationship between turbidity and suspended sediment is nonlinear on an arithmetic plot. Generally about 80% of the variability in suspended sediment concentrations can be explained by simultaneous turbidity measurements. Detailed analyses of data from three sites in Vermont (Kunkle and Comer, 1971) and the five key monitoring stations on the Bull Run municipal watershed near Portland (Aumen et al., 1989) indicated that a single relationship could be used to predict suspended sediment concentrations at each group of closely-related sites.

On the other hand, three watersheds in the Oregon Coast Range showed significant differences in the relationship between suspended sediment and turbidity (Beschta, 1980). There also were significant differences in the suspended sediment-turbidity relationship for different storms at each site. Nevertheless, the pooled turbidity data for each watershed still could account for nearly 80% of the variability in suspended sediment concentrations on that watershed (Brown, 1983).

Turbidity tends to be less sensitive to the sampling location within a stream than suspended sediment, as turbidity is primarily a function of the smaller particles (silts, clays, and colloids). Hence the materials causing turbidity tend to be more evenly distributed within the water column and across the stream cross-section, and grab samples usually are considered to be sufficiently representative. It is recommended that samples be analyzed for turbidity within 24 hours (APHA, 1980), as algal growth can cause an increase in turbidity. In forested areas it is often assumed that water temperature and water quality (e.g., paucity of nutrients) will inhibit or restrict algal growth, but protocols for sample collection and storage should consider this possibility. Sediment flocculation also can cause turbidity values to change over time.

The variability in turbidity among sites and over time generally makes it quite difficult to determine a natural or background level for any specified level of discharge. The natural variation is almost always greater than 10% about the mean for any given discharge, and the variation tends to increase with higher discharges (Brown, 1983). Uncertainty due to instrument differences and analytical errors also amount to approximately 10% (APHA, 1980). The combined uncertainty due to natural variability and measurement errors has important implications both for detecting increases in turbidity due to forest harvest and other management activities, and for enforcing relatively narrow turbidity standards.

Standards

Turbidity standards can be either relative or absolute. Drinking water standards usually are in absolute terms, and current EPA regulations require turbidity in municipal water supplies not to exceed 1 NTU (EPA, 1986b).

Relative turbidity standards have been established in some states. California, for example, specifies that a timber harvest cannot increase turbidity by more than 20% above background. Alaska and Washington allow an increase of 5 NTU for domestic water supplies when the background turbidity is less than 50 NTU, and no more than a 10% increase in turbidity when the background level is greater than 50 NTU (Harvey, 1989). The general criteria for the protection of freshwater fish and other aquatic life is that the depth of the photosynthetic compensation point should not be reduced by more than 10% from the seasonally established norm for aquatic life (EPA, 1986b). As suggested above, the basic problem with enforcing these standards is that background levels are seldom defined and difficult to determine. This suggests that only continuing major violations can be unambiguously identified.

Current Uses

Probably the most common use of turbidity measurements is to monitor the quality of domestic water supplies. More frequent sampling is required as the measured turbidity approaches or exceeds the 1 NTU standard.

Turbidity often is used to monitor the effects of a specific management activity (project monitoring). Typically this involves a comparison of measurements taken upstream (control) and downstream (treated) of a particular project, such as the construction of a bridge, with the presumption that any increase in turbidity is due to that activity. This procedure is particularly effective during low flow periods when the background turbidity is both low and consistent. Assessing the effects during storm periods is considerably more difficult (i.e., less sensitive).

Turbidity measurements provide an indication of the amount of suspended material in the water, but the precise relationship between turbidity and the mass of suspended material depends on the size and type of suspended particles. This relationship must be established for each stream or sampling location, and simultaneous measurements of suspended sediment and turbidity must be made over the full range of expected discharges. In some cases a single relationship may apply at several sites, but this must be based on a careful statistical analysis of the data from each site. The relationship between suspended sediment and turbidity cannot be assumed to be stable over time, as changes in sediment sources or transport processes may alter the relative balance between suspended sediment and turbidity.

The relative ease of measuring turbidity means that it is commonly used for monitoring nonpoint sources of sediment. Suspended sediment tends to be measured in more detailed studies, or when there is a need to estimate sediment loads (e.g., to calibrate or validate a sediment yield model). If an additional uncertainty of ±25% is acceptable, turbidity can be used to estimate suspended sediment concentrations. Estimation of the suspended sediment load requires continuous discharge measurements.

Assessment

Turbidity is relatively quick and easy to measure. Suspended sediment usually is the primary source of turbidity in forest streams in the Pacific Northwest and Alaska. Simultaneous measurements of suspended sediment and turbidity generally result in a relationship that can predict about 80% of the variation in suspended sediment concentrations from measured turbidity values. Thus turbidity can

be used as a surrogate for suspended sediment concentrations. The relative ease of measuring turbidity means that qualitative field observations and synoptic sampling can be used to identify specific sediment sources (source-search methodology discussed in Part I, Section 3.2.3).

Turbidity is regarded by many as being the single most sensitive measure of the effects of land use on streams. This is due partly to the fact that relatively small amounts of sediment can cause a large change in turbidity, and partly to the estimated accuracy of turbidity measurements (approximately±10%) (APHA, 1980; Brown, 1983). Although the variation in turbidity with discharge generally is greater than 10% (Brown, 1983), both the accuracy and variability of turbidity measurements compare favorably with the other sediment parameters (suspended sediment and bedload) as well as the channel characteristics (Chapter 5).

The disadvantages of turbidity are twofold. First, the relationship with suspended sediment must be determined for each site, even though some studies have shown that several sites with similar physical characteristics may have identical relationships. Second, turbidity is highly variable. As in the case of suspended sediment (Section 4.1), turbidity varies according to the discharge; the occurrence of sporadic events such as debris flows, landslides, or the breakdown of log jams; the timing of the sample relative to the season of the year; the time since the last runoff event; and the timing within a storm hydrograph. The range and nonlinear nature of these variations make it very difficult to establish and enforce a narrowly defined turbidity standard for storm events. Narrow turbidity standards are much easier to develop and apply during low flow periods when background levels are consistently low (e.g., a comparison of turbidity levels upstream and downstream of a bridge construction site).

Turbidity measurements are particularly effective in the case of project monitoring (e.g., samples are taken upstream and downstream of a particular management activity).

4.3 BEDLOAD

Definition

Bedload is the material transported downstream by sliding, rolling, or bouncing along the channel bottom (Ritter, 1978). Typically particles >1.0 mm in diameter are transported as bedload, while particles <0.1 mm in diameter are transported as suspended load. Particles between 0.1 and 1.0 mm in diameter can be transported either as suspended load or as bedload depending on the local hydraulic conditions (Everest et al., 1987). Thus even at a single site a particle may be transported as bedload or suspended load depending on the discharge and other hydraulic factors.

Bed material load, a term often confused with bedload, is the transport of particles of a grain size normally found in the stream bed (Linsley et al., 1982). Thus a stream bed comprised primarily of silt and clay particles will have most of its bed material load transported as suspended sediment, while the bed material load of a coarse-bedded stream (e.g., gravels and cobbles) will be transported almost entirely as bedload.

Relation to Designated Uses

Bedload is an important component of the total sediment load of a stream. The proportion of the sediment load transported as bedload varies considerably and cannot be characterized by a simple relationship to suspended sediment load or to discharge (Williams, 1989b).

The amount and size of the bed material, in conjunction with the discharge, slope, and geology, largely determine the overall type and shape of the channel. Wide, shallow channels are characteristic of streams transporting coarse bedload in unconstrained alluvial valleys (Ritter, 1978). As discussed in Sections 5.1-5.2, streams with a high width-depth ratio are more likely to experience high water temperatures that may be detrimental to coldwater fisheries. Streams with coarse bedload tend to have a lower sinuosity than streams that have fine particles as their bed material (Section 5.6.1; Schumm, 1960). Streams with high volumes of bedload and erodible banks often are braided, and the rapid changes in channel location characteristic of braided streams result in continuing high erosion and sediment transport rates. The unstable channels in braided reaches provide relatively poor habitat for salmonids, and the large amounts of sediment transported downstream from braided reaches can adversely affect reservoir storage capacity and other designated uses such as fisheries and irrigation.

Large amounts of easily transported bedload tend to fill in pools and reduce the larger-scale features that are important sources of fish habitat. At very high flows, however, the pools may be scoured (e.g., Campbell and Sidle, 1985).

The type and amount of bedload is very important in determining the amount of microhabitat available for juvenile fish and macroinvertebrates (Section 5.6.1). In general, coarser material provides more habitat space, whereas fine sediments tend to fill up the interstitial spaces between larger particles. Fine sediment is usually defined as particles <0.83 mm in diameter, but some studies have used values of up to 6.4 mm (Everest et al., 1987). The deposition of fine sediment reduces the habitat space for young fish and aquatic macroinvertebrates (Sections 5.6.1, 7.3, and 7.4; Everest et al., 1987).

The deposition of these finer bedload materials (e.g., sand-sized particles) also has been shown to adversely affect gravel permeability and the suitability of the gravel for spawning salmonids (e.g., Everest et al., 1987; Lisle, 1989). A lower permeability usually reduces the concentration of intergravel dissolved oxygen (Section 2.4), and this can be directly related to salmonid spawning success, and

the number and diversity of aquatic invertebrates (Chapman and McLeod, 1987).

As suggested above, the deposition of bedload has an adverse effect on reservoir capacity and can clog up irrigation and shipping channels. Hundreds of millions of dollars are spent in the U.S. each year to remove sediment deposited behind dams and in the lower reaches of rivers and estuaries.

Effects of Management Activities

The effect of forest management activities on the availability and transport of bedload has been shown to range from severe (e.g., Megahan et al., 1980) to no significant difference (Moring, 1975; Sheridan et al., 1984). Part of the observed variation in effects is due to the type and intensity of management. In southwest Oregon, for example, clearcutting was found to approximately double the bedload yield as compared to a control watershed, while patch and selection cuts had no apparent effect (Adams and Stack, 1989). The range of erosion and sediment transport processes operating in the Pacific Northwest and Alaska is another reason why widely different results should be expected from different studies, and why simple generalizations cannot be made about the effects of management activities on bedload (Swanson et al., 1987).

As noted in Sections 4.1 and 4.2, forest harvest can increase erosion rates by generating overland flow on compacted areas, increasing the number of slope failures (e.g., Ice, 1985; Megahan and Bohn, 1989), and increasing the rate of dry ravel and soil creep (e.g., Ziemer, 1984). Alterations in the amount of large woody debris (LWD) in the stream channels will alter the sediment storage capacity in the stream channel (Section 5.7; Megahan, 1982). Removal of LWD, or a reduced rate of recruitment of LWD into the stream channel, can result in an apparent increase in sediment yield at the mouth of the basin (Megahan, 1982), even though there may be no net change in the rate of sediment delivered to the stream channel from upslope.

Road construction and road maintenance can increase the amount of bedload by creating areas prone to surface runoff (Reid and Dunne, 1984), altering slope stabilities in cut and fill areas (e.g., Megahan, 1978), and altering drainage patterns in ways that tend to increase the number of landslides and debris flows (e.g., Megahan et al., 1978; Megahan and Bohn, 1989). Similarly, grazing can increase the amount of overland flow and decrease bank stability (Section 5.8; Gifford, 1981). Sand and gravel extraction within the stream channel will alter the channel hydraulics and probably cause a short-term increase in bedload transport until the stream re-establishes a stable channel. Longer-term effects of sand and gravel extraction are difficult to predict.

The material eroded or detached by these different hillslope erosional processes must then be delivered to the stream channel and transported by the stream before it can be measured as bedload. Often significant amounts of material can be stored in the channel (Dietrich et al., 1982). In streams draining the Idaho batholith, for example, 15 times more sediment was stored in the channel than was delivered out of the basin on an annual basis (Megahan, 1982). When evaluating the impact of management activities on bedload, one must also consider whether the material is composed of silt- and clay-sized particles, which probably will be transported as suspended sediment, or coarser particles, which will be transported as bedload.

Extensive studies on the South Fork of the Salmon River in Idaho have attempted to link the effects of forest management and road building to an increase in bedload and the quality of fish habitat. In this basin the combination of management activities, erodible soils, and severe storms has resulted in extensive sedimentation. The large amounts of bedload reduced pool depths and literally buried many of the prime salmonid spawning and rearing areas with sand (Megahan, 1980; Box 3, page 19). In other parts of the Pacific Northwest, studies have documented increased amounts of fine sediment in the bed material in response to forest harvest and road-building (Section 5.6.1; Cederholm et al., 1981; Scrivener, 1988). However, very few published studies have attempted to monitor changes in bedload transport rates due to forest management activities, and then relate these changes to the designated uses of the water body being monitored. The paucity of such studies has strong implications with regard to the relative utility of monitoring bedload transport rates.

Measurement Concepts

The measurement of bedload must be regarded as difficult. Sampling devices disturb the flow in the vicinity of the sampler, and this biases the sample (Guy and Norman, 1970; Emmett, 1980). The most common bedload sampling device, the Helley-Smith sampler, consists of a flared rectangular orifice with an attached mesh bag. The sampler is placed on the stream bottom with the opening facing upstream for a specified time, and the sediment caught in the mesh bag is dried and weighed to get a transport rate in mass per unit time per unit stream width (Helley and Smith, 1971). The most commonly used design has a 76-mm (3.0-inch) square opening and a mesh size for the sample bag of about 0.25 mm. This has been reported to have a catch efficiency of about 1.0 for particles from 0.5-16 mm in diameter (Emmett, 1980). Sampling of larger bedload particles requires a larger sampler, and the catch efficiency is less well known.

Bedload transport rates vary across the stream cross-section, so representative samples should be taken at regular intervals across the stream (Emmett, 1980). Numerous studies, however, have shown that bedload moves in irregular sheets or waves (e.g., Beschta, 1981; Reid and Frostick, 1986). This can be due to migrating dunes or bedforms, and to unpredictable events, such as the breakup of a stream

armor layer, the release of sediment stored behind channel obstructions (e.g., Megahan 1982), and sudden inputs of sediment (Swanson et al., 1987; Sidle, 1988). Several studies indicate that bedload transport tends to be higher on the falling limb of the hydrograph (i.e., declining discharge) than on the rising limb of the hydrograph (increasing discharge). This is the reverse of the usual hysteresis effect for suspended sediment, and it is attributed to the initial resistance of the surface armor layer to entrainment (Reid et al., 1985; Lisle, 1989). Thus bedload samples taken from the same location at constant flow can be expected to vary greatly over relatively short time periods, and sampling should be conducted over several sediment transport "cycles" (Emmett, 1980). Under normal field conditions an accuracy of no better than 50-100% can be assumed (e.g., Lisle, 1989). The temporal distribution of bedload transport in any given year will vary according to the size of the bed material and the flow regime of the stream in question, but in most streams the majority of bedload will be transported only during the two or three largest flows in a particular year.

In some cases the long-term sediment transport rate can be estimated by measuring the amount of sediment that accumulates in a lake or other sediment trap (Foster et al., 1990). Such estimates may need to be adjusted by the trap efficiency, which is a function of the residence time of the inflowing water (Barfield et al., 1981). This procedure generally does not allow separation of bedload and suspended load.

The difficulty of accurately measuring bedload has led to the development of numerous equations to predict bedload transport. However, these equations are seldom able to predict observed transport rates over the entire particle-size range found in natural streams (e.g., Reid and Frostick, 1986). Flume data are often used to develop and validate these transport equations, but they then have to be extrapolated to field conditions.

Standards

No standards have been established or proposed for bedload, and this is probably due to the difficulty of measuring and evaluating bedload transport.

Current Uses

In some monitoring projects a set of bedload samples is taken during selected field visits. Bedload transport, however, is highly dependent on stream discharge and is less frequent than suspended sediment transport. Thus virtually all of the annual bedload transport occurs during peak snowmelt or a few of the largest runoff events. While this greatly shortens the period of sampling, these few events must be intensively sampled if annual bedload transport is to be estimated. Williams (1989a, 1989b) found no consistent relationship between bedload, discharge, and suspended load, and concluded that the concept of a constant bedload proportion is not generally valid.

In streams that are not heavily armored, there may be some value in occasionally measuring bedload transport at different cross-sections during moderately high flows. These measurements may indicate the flow at which the bed material begins to move, and this provides a useful check on theoretical estimates based on depth, velocity, and particle size. Occasional field measurements also will help to understand the relative balance between suspended sediment load and bedload for the sampled discharges. This in turn may help indicate the relative sensitivity of the stream system to different types of sediment inputs.

Relatively few monitoring studies can afford to intensively sample bedload. As a result, estimates of bedload transport cannot be made except in cases where a downstream trap is available that can be surveyed on a regular basis. The high year-to-year variation in bedload transport, particularly for coarser bed materials (e.g., Sidle, 1988), suggests that a relatively long-term record is needed to obtain a reliable estimate of bedload transport rates. Less frequent samples are useful only as crude indicators and generally should be interpreted qualitatively rather than quantitatively.

Assessment

One can argue that some data are always better than no data, but in the case of bedload it is questionable whether a limited amount of quantitative data has any real value for estimating bedload transport (Williams, 1989b). Unless the stream is intensively sampled during high flow events, or a trap exists where sediment accumulations can be periodically measured, annual bedload transport should not be estimated.

As suggested above, occasional bedload samples may be helpful in gaining a qualitative assessment of stream behavior. Typically most field investigations are conducted during low flow periods and require a series of geomorphologic clues to develop an appreciation for stream condition, sediment transport capacity, and sensitivity to increased sediment load. Bedload samples during high flow events, when separated by particle size and combined with discharge data, may help this interpretation process. Bedload data rarely provide unique, quantitative information for anything more than very crude model verification or forest planning, but they can provide some additional insight into stream behavior.

Another problem with measuring bedload transport rates is the difficulty of directly relating specific bedload transport rates to adverse effects on the various designated uses such as salmonid habitat. Often the effects of bedload on the designated uses can be more directly assessed by monitoring parameters such as cobble embeddedness, residual pool depths, pool-riffle ratios, or cross-section profiles. Although these parameters all have their own draw-

backs, and it may be difficult to link observed changes to specific management activities, at least such measurements can be directly related to many of the important designated uses of forest streams. Most of the channel characteristic parameters also have the advantage of being considerably easier to measure than bedload transport, and these are the subject of the following chapter.

5. CHANNEL CHARACTERISTICS

INTRODUCTION

The parameters reviewed in this chapter relate to the shape of the stream channel, the structural features within the stream channel, and the stability of the stream banks. These channel characteristics can be monitored on different spatial scales and from different perspectives. For example, bed material particle size and embeddedness evaluate the surface of the streambed on a scale of a few centimeters, whereas a thalweg profile evaluates the topography of the deepest part of the streambed on a scale of tens or hundreds of meters. Measurements of habitat type (e.g., pools, riffles, etc.) were pioneered by fish biologists and are used to evaluate the quality of fish habitat, but these measurements are functionally related to the parameters that might be used by fluvial geomorphologists (e.g., residual pool depth or the number of debris dams caused by large woody debris).

Most of the characteristics of stream channels that might be used for monitoring are controlled by the same basic set of interacting factors. Among the most important of these are the amount and size of sediment, the duration and size of peak flows, slope of the valley bottom, valley bottom width, steepness of the sideslopes, and the local geology. Some of these factors can be considered constant for a given site, while the factors that do vary (discharge and sediment) are relatively difficult to monitor (Chapters 3 and 4). Stream channel characteristics may be advantageous for monitoring because their temporal variability is relatively low, and direct links can be made between observed changes and some key designated uses such as coldwater fisheries.

The importance of these controlling factors suggests that many of the channel characteristics will have a similar response to management activities. Some of the parameters which are most closely related include channel cross-sections (Section 5.1) and channel width/width-depth ratio (Section 5.2); pool parameters (Section 5.3) and thalweg profile (Section 5.4); and the three parameters relating to bed material (particle size, embeddedness, and surface vs. subsurface bed material particle size; Section 5.5). In most cases it is not necessary to monitor each of these closely related parameters, and the selection among these monitoring parameters will depend upon the particular combination of management activities, designated uses, and site-specific conditions. General recommendations are difficult because relatively few studies have used channel characteristics as the primary parameters for monitoring management impacts on streams.

The relatively low temporal variability of channel characteristics must be balanced against (1) the potentially large spatial variability, and (2) the problem of separating man-induced changes from changes due to natural events. Proper statistical design can help alleviate both of these considerations, and the much lower frequency of sampling will allow more sites or more parameters to be measured. In many cases a combination of several channel parameters may be the best approach to evaluate and understand observed changes in the stream channel.

5.1 CHANNEL CROSS-SECTION

Definition

A channel cross-section is a topographic profile of the stream banks and stream bed along a transect perpendicular to the direction of flow. Cross-sectional data are obtained by measuring distance and surface elevations along the designated transect or cross-section. The endpoints of the cross-section are arbitrary, but they should extend at least above the estimated bankfull stage and preferably beyond the current floodplain. If change over time is to be monitored, the elevation data must be related to a permanent benchmark.

Cross-section data are needed to calculate discharge using any of the velocity-area methods (Buchanan and Somers, 1969). Cross-sections often are used as the sampling transect for other instream parameters such as bed material particle size (Section 5.6.1), embeddedness (Section 5.6.2), and the type and amount of large woody debris (Section 5.7).

A series of cross-sections referenced to a single benchmark is useful to determine the precise slope of the stream channel. Channel slope is a key parameter in most hydraulic calculations and for stream classification.

Relation to Designated Uses

A cross-section of the channel and adjacent floodplain is one of the key pieces of information necessary to predict the velocity and water surface elevation during high flow events. Such predictions are needed for a variety of engineering and management purposes, including structural design, estimation of flood heights, and the stability of channel protection measures.

For these types of engineering purposes, cross-section data typically are collected at a single point in time. At best such data can provide only a qualitative indication of channel condition.

Monitoring of changes in the channel cross-section can provide important insights into channel stability, bank stability, and the relative balance between sediment (particularly bedload) and discharge (Beschta and Platts, 1986). Widening of the stream channel, filling in of the channel thalweg (the deepest portion of the channel), increasing bed elevation (i.e., channel aggradation), and declining cross-sectional area all indicate an excess of sediment. Net deposition of sediment usually results in more extreme stream temperatures, a decrease in the amount and quality of fish cover, a change in the quality of the spawning habitat, a possible reduction in habitat space for algae and macroinvertebrates, increased bank erosion, and an increased likelihood of flooding (Section 4.1).

Channel incision or bed erosion (channel degradation) usually indicates a reduction in coarse sediment inputs or an increase in sediment transport capacity due to higher peak flows. This can have beneficial or adverse effects depending upon the initial conditions and the designated use(s). Channel incision will lead to bank steepening and bank instability, and this will increase the sediment load. Bank instability also will lead to a toppling of the riparian woody vegetation immediately adjacent to the stream channel, which can trigger a series of secondary effects (Section 6.2). On the other hand, if the channel already has been subjected to increased sediment loads from previous management activities, channel incision may represent a return to "natural" conditions and an improvement in habitat quality and channel capacity (e.g., Megahan et al., 1980).

Response to Management Activities

The shape and area of the channel cross-section can change in response to a variety of management activities. Management can alter the size or frequency of peak flows (Section 3.1) and the sediment load (Chapter 4), and these are likely to affect the shape and area of the channel cross-section. A decline in bank slope may be due to grazing impacts. Rapid infilling and an increase in the width-depth ratio suggests an excess of coarse sediment. Erosion at the toe of the bank may lead to a slumping of the oversteepened bank, and these changes can be quantified by systematically monitoring selected cross-sections.

In each case additional information on management activities and natural events should be collected. For example, the cause of infilling could be either several years of below-average rainfall or an upstream landslide. In northern California and parts of the Northwest, an apparent downcutting in certain stream channels is actually part of the long-term recovery from the large sediment deposits associated with the extreme 1964 flood (Lisle, 1982).

Measurement Concepts

Cross-sections are surveyed by establishing a line perpendicular to a stream and measuring bed surface elevations either at regular intervals or at pronounced changes in slope. If the cross-section is not perpendicular to the stream channel and flow direction, errors will accumulate in the estimates of cross-sectional area and discharge. Cross-section data always should be plotted for error-checking and improved visualization of channel form.

Typically a cross-section is measured by a two-person crew with surveying equipment. However, one person can survey a cross-section by stretching a tape across the stream and then measuring the height of the tape above the ground surface. Some investigators have found this latter technique to be more efficient (Platts et al., 1983).

Often a series of cross-sections are necessary to characterize a stream reach, establish transects for sampling other parameters, and provide quantitative data for statistical analysis. Groups or clusters of cross-sections can be located by random sampling, stratified random sampling, or systematic placement around random samples (Part I, Chapter 3). Stratified random sampling can be effective in reducing variability, decreasing sample size, or increasing the ability to detect change if the strata are properly chosen and the user has some prior information on the types and variability of the strata. Either of the two random sampling techniques are acceptable provided the number of samples is large enough to meet the statistical requirements (Platts et al., 1983). Data from cross-sections can be grouped by habitat type (Section 5.5) to determine general trends.

Standards

No standards for channel cross-sections have been established or proposed.

Current Uses

The most common reason for collecting cross-section data is to calculate discharge using the standard velocity-area technique (Buchanan and Somers, 1969). Data from multiple cross-sections are used to evaluate fish habitat conditions, estimate net sediment transport within a particular reach, and evaluate changes in channel morphology (e.g., width-depth ratio, bank slope, and bankfull depth). Certain other parameters, such as bed material particle size and embeddedness, can be properly interpreted only if they are referenced to a particular location along a thalweg profile or channel cross-section.

Cross-section data have been an important component of monitoring the South Fork of the Salmon River (Box 3, page 19) and the Silver Fire Recovery Project (Box 7, page 57). In many other monitoring projects, cross-section data have been collected but have not been analyzed to determine the specific changes occurring over time. The ready availability of computer software programs and digitizing tables means that comparative analyses can be done more quickly than in the past. Reference cross-sections are being established by the Timber-Fish-Wildlife Ambient Monitoring Program in Washington and by the U.S. Forest Service.

Assessment

Stream cross-sections provide a quick and useful visualization of the stream channel. Repeated measurements of the same cross-section is a relatively simple means to monitor changes in the stream channel. Sampling locations for other monitoring parameters often are established on the basis of reference cross-sections.

The sensitivity of a cross-section to change is highly dependent on a variety of site factors. Bedrock can limit scour or lateral migration. In steeper reaches, where the stream has a high sediment transport capacity, there may be no net deposition despite an increase in sediment load. Conversely, in downstream alluvial reaches the channel cross-section may be relatively responsive to changes in both the sediment load and the size of peak flows. This suggests that a series of cross-sections may be needed to assess the overall patterns of channel change within a catchment.

The primary problem with monitoring cross-sections is that it may be very difficult to determine the cause of an observed change. A channel cross-section represents an integrated response to natural events, the physical environment, and management impacts. Separation of these factors requires several different approaches. First, cross-sections should be monitored over a relatively long time period, as short-term changes resulting from unusual climatic events can mask a quite different overall trend. Second, data on other parameters, such as bed material particle size or riparian vegetation, are necessary to fully characterize and understand any observed changes in channel morphology. Finally, the data on channel cross-sections must be put in the context of a broader watershed assessment, and this should include data on the type and location of management activities, watershed characteristics, and the historical climate.

In summary, cross-section data are most useful if combined with other monitoring parameters. Cross-section data alone may be difficult to relate directly to the designated uses of the water body of concern. A determination of channel aggradation or degradation, for example, may permit inferences to be made about certain designated uses such as wildlife or fisheries, but are not a direct measure of these uses and may not indicate the cause of an observed change. On the other hand, channel cross-sections are relatively easy and inexpensive to measure, particularly in smaller streams. Thus a combination of channel cross-section data with other parameters more closely linked to the key designated uses (e.g., spawning habitat) can provide the basis for a relatively powerful and inexpensive monitoring procedure.

5.2 CHANNEL WIDTH/WIDTH-DEPTH RATIOS

Definition

Sediment accumulation in the stream channel reduces stream depth. To maintain the same channel capacity, there usually is a corresponding increase in stream width. These interrelated changes provide the basis for two geomorphic parameters that can be used for monitoring purposes—stream width and the width-depth ratio.

Both stream width and stream depth have to be defined with regard to a certain discharge. This discharge can be specified in absolute terms (e.g., 30 cubic feet per second), in geomorphic terms (e.g., bankfull), or in terms of recurrence interval (e.g., a 5-year event). Because streams almost always are several times wider than they are deep, a small change in depth can greatly affect the width-depth ratio. One must also specify whether the depth is the average depth for the cross-section or the maximum (thalweg) depth.

An alternative to measuring channel width is to monitor the width of the riparian canopy opening, and this approach is reviewed in Section 6.1.

Relation to Designated Uses

A decrease in channel depth and an increase in channel width can have major adverse effects on the biological community. A decrease in depth tends to reduce the number of pools (Beschta and Platts, 1986), and this will reduce certain types of fish habitat. An increase in stream width will lead to an increase in net solar radiation and higher summer water temperatures (Beschta et al., 1987). The combination of shallower pools and increased solar radiation can greatly affect the suitability of the stream for coldwater fisheries. An increase in stream width and an increase in light penetration is likely to increase primary production, although this may be partly offset by a reduced input of organic debris into the aquatic ecosystem from the riparian zone (Gregory et al., 1987).

An increase in channel width is achieved through bank erosion and a corresponding increase in sediment inputs into the stream channel. An increase in bank erosion is particularly important because the sediment is delivered directly into the stream channel (Section 5.8). The adverse effects of an increased sediment load were reviewed in Chapter 4.

An increase in the riparian canopy opening due to an increase in stream width can have a series of adverse biological effects. Such an increase is likely to reduce the amount of riparian vegetation, and this will reduce the ability of the riparian zone to capture nutrients and sediment (Section 6.2). The riparian zone is also a major source for large woody debris, an important element in pool formation and habitat diversity in most forested streams in the Pacific Northwest and Alaska (Section 5.7).

Response to Management Activities

Forest harvest, road building, road maintenance, and other management activities often increase the amount of sediment delivered to the stream channel. Usually an increase in coarse sediment will lead to an accumulation of sediment in the deeper parts of the stream channel. If the runoff remains unchanged, an unconstrained stream generally responds by increasing its width (e.g., Lisle, 1982; Grant, 1988). Although the magnitude of this increase in width will be affected by the valley shape and the bank materials, Lisle (1982) observed increases in width even in constrained, non-alluvial materials. Thus changes in width or the width-depth ratio can be used as an indicator of a change in the relative balance between the sediment load and the sediment transport capacity.

Grant (1988) noted that an increase in channel width also could result from an increase in the size of peak flows. As shown in Section 3.1, increases in the size of peak flows due to forest harvest generally are small except in areas subject to rain-on-snow events. This additional mechanism for channel widening does not preclude the use of channel width as a monitoring technique, but it does suggest that additional data are required to understand the cause of any observed changes. Harvest of the riparian vegetation also can decrease bank and channel stability and thereby initiate a cycle of bank erosion and channel widening (Section 6.2).

Measurement Concepts

The determination of channel width and channel depth is problematical because both parameters are flow-dependent. Depth tends to increase with flow more rapidly than width (Dunne and Leopold, 1978; Leopold and Maddock, 1953), but this relationship may not be constant at a given cross-section. A stream with a wide, flat floodplain, for example, will experience a sudden increase in width when the flow overtops the banks and spreads across the floodplain. Thus the monitoring of changes in width and depth should be done at specified discharges and locations. A geomorphically based discharge, such as active channel width or bankfull width, is most commonly used but may be relatively subjective. The resulting uncertainty must be taken into account when drawing inferences from the data.

Cross-section location will affect the width-depth ratio and, as noted in Section 5.1, the sensitivity to change. For example, stream width and width-depth ratios are likely to differ across riffles, sharp bends, and pools. This variation can be minimized by measuring widths and depths at a consistent channel form such as straight riffle reaches, using average depth rather than maximum depth, or by using average values obtained from several different cross-sections.

The sensitivity of stream width and width/depth ratios to management impacts and natural events will vary with stream type and location. A bedrock stream in a steep, V-shaped valley will not alter its width in response to an increase in sediment load as easily as a stream in a wide valley with unconsolidated alluvial sediments. Channel shape is also affected by the relative proportions and absolute amounts of bedload and suspended load (e.g., Schumm, 1960). Streams with cohesive materials tend to have narrow, deep channels, while streams in a sandy or other non-cohesive substrate tend to be wide and shallow.

Standards

No standards have been set or proposed for changes in stream width or width-depth ratios.

Current Uses

Although a considerable amount of cross-section data can be obtained from gaging stations, stream inventories, and other studies, channel width has not been extensively used as a monitoring technique. Powell (1988) documented the increase in stream width that occurred in both the careful and the intense logging treatments on Carnation Creek in

coastal British Columbia. Channel width and depth data also have been collected in conjunction with the intensive, long-term monitoring effort on the South Fork of the Salmon River (Box 3, page 19; Torquemada and Platts, 1987).

Present efforts by agencies such as the U.S. Forest Service to inventory fish habitat and stream channel condition should generate a large amount of stream width and width-depth data. It remains to be seen how well these particular parameters can define stream condition and monitor management impacts.

Assessment

On-the-ground measurements of channel widths and width-depth ratios have the potential of being relatively sensitive indicators of changes in the size of peak flows and sediment yields. Channel width and width-depth ratio can be related to the value of streams for fish and recreation.

Defining channel width and depth in the field is not a trivial problem. For this reason it is best to monitor channel width at a series of cross-sections. Use of geomorphic indicators such as bankfull width or active channel width must be done with great care, as these tend to be subjective and a major runoff event can alter the channel cross-section and make identification of bankfull features questionable. Determining width and depth at a standard discharge may be logistically difficult unless it is done at an existing gaging station. The problem with using gaging stations as monitoring locations is that they usually are placed at geomorphically stable locations and are relatively insensitive to management-related changes in channel form.

Measuring channel width or width/depth ratios also suffers from the same basic limitation as any other instream measure—namely, that it does not provide any information on the cause of an observed change. Hence monitoring data must be combined with information on management activities, storm events, and sediment sources (e.g., roads, debris flows, landslides, or a breakdown of debris dams). As noted earlier, one also has to put the changes observed from a relatively short-term monitoring project into the context of larger changes such as extreme floods or major sediment inputs. Only with this additional information can the effects of forest management begin to be deciphered.

Finally, the magnitude and rate of change in channel width and width-depth ratio will depend on factors such as the slope of the stream, the shape of the valley bottom, the bank and bed materials, and the recent flood history. Although this may make it difficult to establish specific standards, it should not mask general trends. These considerations also indicate that long-term measurements at various locations within the watershed are needed for adequate monitoring.

5.3 POOL PARAMETERS

Definition

Pools can be defined as sections of the stream channel that have a concave profile along the longitudinal axis of the stream, or as areas of the stream channel that would contain water even if there were no flow. This means that the maximum depth of pools is deeper than the average thalweg depth, and water velocities at low flows often are lower than the average velocity. Pools are an important component of the aquatic habitat, and they can be classified and measured in several different ways.

Pools usually are classified by the process that created the pool (e.g., undercut bank, debris dam, beaver dam, plunge pool, etc.). This classification is useful for evaluating the abundance and type of fish habitat (Bisson et al., 1982), although the various categories of pools and other habitat types have not been standardized (Section 5.5; Platts, 1983). Nevertheless, the number and type of pools in a particular reach could be enumerated, and changes over time could be monitored.

More commonly the depth, residual depth, volume, or area of pools are measured, and these measurements can be used as monitoring parameters. Pool depth can be either average depth or maximum depth. Residual pool depth refers to the depth of the pool below the downstream lip of the pool (i.e., the depth of the water which would be trapped in the pool if there was no discharge) (Lisle, 1987). Pool area refers to the total surface area of the pool. Both pool depth and pool area will vary with discharge, whereas residual pool depth is not discharge-dependent.

Relation to Designated Uses

Pools are an important morphological feature in stream channels and an essential type of fish habitat. In general, a variety of pool types are needed to provide the range of habitat needed by different species and age classes of fish. Slow-moving dammed or backwater pools may be necessary for salmonid survival under harsh winter conditions. Deep undercut pools may provide protection from high temperatures. Young fish may require shallow, low-quality pools to avoid predation. Particularly in smaller streams, pools provide the majority of the summer rearing habitat (Beschta and Platts, 1986). Pools also may be important sites for recreational activities such as fishing and swimming.

Response to Management Activities

Those pools characterized by low flow velocities (e.g., backwater or dammed pools) are particularly susceptible to infilling with sediment. Hence the depth, area, or volume of

these pools can serve as a relatively sensitive indicator of changes in the coarse sediment load due to forest harvest, road building and maintenance, mining, or other management activities. On the South Fork of the Salmon River logging and road maintenance caused an influx of sand-sized material that filled in many of the prime salmonid spawning and rearing areas (Megahan et al., 1980).

Changes in pool area, pool volume, or residual pool depth also can be caused by changes in the features that create pools. Thus a reduction in the input of large woody debris may lead to a reduction in the number and size of pools (Section 5.7). Similarly, a change in the size or frequency of peak flows will alter the ability of the stream to transport coarse sediment, and this may alter pool measurements.

The total area, depth, or frequency of pools may not always be a reliable indicator of adverse management effects. Streams immediately downstream of active glaciers, for example, usually are braided and have little or no pool areas. Landslides, debris flows, and other mass movements typically result in a loss of pool area and volume, and these pulsed inputs of sediment may or may not be triggered by management activities (Swanson et al., 1987).

Measurement Concepts

Pool depth, pool area, and pool volume are all direct physical measurements, and they are relatively simple to make in small streams. Recent publications have encouraged the use of visually estimating the width, depth, or area of pools within a stream reach, and then adjusting these visual estimates for any systematic bias by measuring a certain percentage of the pools (Hankin and Reeves, 1988). In larger streams with deeper pools, direct measurements are considerably more difficult. Also, a series of conceptual problems in making pool measurements must be considered before embarking on a classification or monitoring program.

First, it may be difficult to determine exactly what constitutes a pool. Large, still pools are easy to classify, but the change from pools to runs or glides is one point on a continuum. Platts et al. (1983) found a consistent observer bias when measuring pool areas along stream cross-sections. This consistent bias resulted in a relatively narrow 95% confidence interval for the data (±10%), but poor year-to-year accuracy and precision.

A second problem associated with pool measurements is that pool depth, pool area, and pool volume are all flow-dependent. An increase in stage will increase the value of these parameters. Although this may not be a problem in streams with a consistent summer baseflow, it does mean that stage or water depth must be recorded and taken into account when analyzing the data. The advantage of residual pool depth is that it is independent of discharge (Lisle, 1987).

Similarly, the classification of pools and other habitat types is stage-dependent, but this fact is often ignored (Section 5.5). At higher flows a pool may become a run, or a pocket water may become a riffle. Hence any summary statistics on pool-riffle ratios or the frequency of pool types also must consider the discharge at the time the data were collected. For this reason comparisons between surveys must be done with extreme caution.

Standards

No standards for any pool parameters have been established or proposed.

Current Uses

Most surveys of fish habitat or stream channel condition have utilized some measure of pool area, length, depth, or volume. Many of these surveys also identify the primary cause of each pool. These data are then used to generate summary statistics on the pool-riffle ratio, pool area, or pool volume per unit length of stream channel. The expectation is that subsequent surveys should be able to determine whether substantial shifts have occurred in these values. Alternatively, one could monitor changes in individual pools, but this approach assumes that the pool-forming structure is constant in time. Studies of woody debris in streams indicate that the larger pieces are relatively stable (Sedell et al., 1988), but it would be prudent to monitor at least several pools of as many different types as possible.

Pool parameters probably are most useful in alluvial channels. Studies of stream channel development following the Mount St. Helens eruption indicate that in many reaches a riffle-pool geometry developed after only a couple of years (Meyer and Martinson, 1989). This suggests that pools could be used for monitoring even under relatively high sediment loads. Pool parameters are unlikely to be useful in bedrock channels that are regularly scoured by high flows.

Assessment

In many streams, pool parameters have considerable potential for monitoring. Decreases in pool depth or pool volume may be relatively sensitive indicators of logging-induced changes in the coarse sediment load or the size of peak flows. Since pool parameters have not been extensively monitored in the past, there is little documentation to guide the selection of a particular parameter. Residual pool depth does have the advantage of being independent of discharge. Residual pool depth also may be the most sensitive pool parameter, as an increase in coarse sediment is likely to first affect pool depth. Monitoring pool parameters will be most useful in low or moderate gradient streams in alluvial valleys (Everest et al., 1987).

To be useful, any monitoring of pool parameters should be combined with data on the pool-forming features. Logging in or near the riparian zone, for example, may alter the type and amount of large woody debris in the stream channel, and this will directly affect the number and size of pools. This suggests that the sample size should be large enough to allow for random changes in pool-creating structures, or the pools should be stratified by pool type.

Pool measurements are most likely to be useful when combined with discharge and other morphological data. Bed material particle size (Section 5.6) can be an extremely useful parameter in interpreting the cause and significance of a change in pool depth or pool volume. Flood history and local discharge data are important because large storms can reduce the size or number of pools, and this effect must be distinguished from forest management activities. Additional long-term data are needed to better assess the value of pool parameters for monitoring, but pool parameters promise some significant conceptual and practical advantages in monitoring forest activities.

5.4 THALWEG PROFILE

Definition

The "thalweg" is defined as the deepest part of the stream channel at any given cross-section. A thalweg profile refers to the topographic variation of the thalweg along the stream axis (i.e., in the upstream-downstream direction). This can be measured with regard to the water surface or surveyed against a fixed elevation. A survey of the thalweg with regard to a benchmark elevation also can be referred to as a longitudinal profile. Sometimes, however, a longitudinal profile can refer to a profile along the streambank or water surface. Thus thalweg profile and longitudinal profile often are synonymous, but this may not always be the case.

Elevation data from a surveyed thalweg profile can be used to calculate an average channel gradient. Thalweg profile data show the variation in bed structure (e.g., pools, riffles, etc.) along the surveyed reach. In particular, a thalweg profile can accurately delineate pools along the main channel and be used to determine residual pool depth (Section 5.3). Both a cross-section (Section 5.1) and a thalweg profile can provide data on the overall degradation/aggradation of the stream channel, but only a thalweg profile can provide quantitative information on the structure and gradients along the stream axis. The length of the thalweg profile also can be compared with the length of the valley floor to yield the thalweg sinuosity. In most cases the thalweg sinuosity will be similar to the channel sinuosity.

Relation to Designated Uses

The average gradient as determined by a thalweg profile is an important criterion for classifying streams. The channel gradient also is needed for a wide variety of hydraulic calculations and models, including water surface profiles and sediment transport capacity. Local gradients are important for estimating shear stress and small-scale hydraulic behavior.

Thalweg profiles provide detailed and unambiguous data on pool depth and pool length. These pool parameters can be directly related to fish habitat value (e.g., Bisson et al., 1982). Changes in flow velocities and stream depths due to changes in the bed profile will affect the number and type of aquatic organisms. An estimate of channel sinuosity is useful for stream classification (e.g., Rosgen, 1985; Cupp, 1989), and for helping to evaluate one of the ways in which energy is dissipated in streams (e.g., Schumm, 1977).

Effects of Management Activities

Changes in sediment load or peak runoff can affect the overall elevation of the thalweg profile through aggradation or degradation, and alter the structure and habitat types along the profile (Beschta and Platts, 1986). More specifically, an increased sediment load can affect local gradients by filling in pools and by reducing the gradient within steep riffles (Sullivan et al., 1987). As discussed in Section 5.3, pool infilling can be a relatively sensitive indicator of adverse management impacts. A decline in sediment tends to result in channel incision, and this has been observed downstream of newly built dams (e.g., Shen and Lu, 1983; Bradley and Smith, 1984) and after a moratorium on timber harvest (Megahan et al., 1980).

A change in the size of peak flows also can be expected to affect the thalweg profile by altering the sediment transport capacity. An increase in peak flows will tend to increase the stream channel width and depth (e.g., Schumm, 1977), but the interactions among bed material transport, bank erosion, sediment inputs, and discharge often make it difficult to predict the precise effect of a change in one factor on the change in other factors. Beschta and Platts (1986) suggest that stream channel morphology is affected more by management-induced changes in sediment than management-induced changes in flow.

Measurement Concepts

A thalweg profile is a relatively simple monitoring technique, and it is relatively inexpensive to obtain in small streams. Surveying equipment is needed to obtain sufficient accuracy. In areas with dense riparian vegetation, the task becomes more difficult because of the problems associated with obtaining a clear line of sight. Surveying a thalweg profile can be difficult on bends, as the thalweg usually

coincides with the area of greatest velocity and may lie beneath an undercut bank. In larger streams a boat and other equipment may be needed to accurately locate and measure the thalweg profile.

No standard length exists for a thalweg profile, but a general rule of thumb is that it should extend for approximately 20-30 channel widths or 2-3 meander segments. In general it should include at least several distinct pools, but the exact location and length will depend upon the objectives of the monitoring and the expected changes in stream channel morphology.

The length of a thalweg profile, when divided by the equivalent length of the valley floor, yields the thalweg sinuosity. The thalweg sinuosity will be similar to, or may slightly overestimate, the channel sinuosity. For short profiles it may be possible to directly measure the valley floor length. Longer thalweg profiles should start and stop at easily defined locations such as bridges so that the valley bottom length can be measured from topographic maps. Thalweg profiles longer than 2-3 meander lengths should be used if an accurate estimate of sinuosity is needed.

Standards

At present no standards or regulations exist regarding a thalweg profile. The state of Idaho, however, is considering the use of thalweg profiles and residual pool depths to monitor sediment production.

Current Uses

In the past thalweg profiles have been measured primarily in the context of research on stream hydraulics and fish habitat. Relatively little long-term monitoring data are available. Nevertheless, surveyed thalweg profiles are attracting increasing interest because of their relative sensitivity to increased sediment inputs, and their ability to quantitatively and unambiguously assess changes in stream channel morphology.

The same studies that support the use of pool parameters as indicators of management effects also can be used to support the use of thalweg profiles. As with any monitoring technique, thalweg profiles are subject to the problem of separating man-induced impacts from natural changes. However, a thalweg profile may have some advantage in that it relies on detailed measurements in a particular location. This enables one to separate individual changes in the stream profile—e.g., the breakdown of a particular debris dam—from the general trend.

Assessment

Thalweg profiles are a specific technique for assessing certain types of changes in stream channel morphology over time. A thalweg profile is complementary to channel cross-sections in that it evaluates changes along the length of a reach, and it offers a possibly more rigorous approach to monitoring the frequency, depth, and length of pools. On the other hand, a thalweg profile cannot provide as much detail on all the different habitat types which are of concern to fisheries biologists (e.g., pocket water, runs, etc.) and which might occur along a typical thalweg profile. Thalweg profiles also can yield data on sinuosity and gradient; both of these are useful for classifying streams and a variety of other purposes.

The disadvantages of thalweg profiles are similar to the other parameters used to monitor channel characteristics. One major problem is how to link an observed change in the stream channel with a particular management activity. This problem is particularly acute for the channel morphology parameters, as their values are the integrated result of a large number of interacting processes. This is why a combination of parameters may be needed to properly evaluate the changes due to management activities and determine the possible cause(s).

Another disadvantage is the problem of setting a threshold or standard for allowable change. In the case of thalweg profiles, one should not just look at an overall change in the gradient, but attempt to interpret all of the smaller changes in bed slope and pool size. Both qualitative and quantitative evaluations may be needed, as streams vary greatly in their sensitivity and response to management impacts (Sullivan et al., 1987). The more recent stream classification schemes (e.g., Cupp, 1989; Frissell, 1987; Rosgen, 1985) may help to interpret thalweg profile data by stratifying the data according to stream type. This will facilitate a comparison among streams, and thereby help to determine the expected range of variability for a particular type of stream.

5.5 Habitat Units

Definition

Most stream reaches in forested areas of the Pacific Northwest encompass a variety of channel features that include different types of riffles and pools. Each of these features provides different habitat values for different fish species at various life history stages. These channel features are referred to as channel units, habitat types, or habitat units. The term habitat unit is used here because it emphasizes the ecological importance of these channel features, and it implies an analysis on a unit-by-unit basis. Habitat type refers to the basic classification system used to delineate individual channel or habitat units.

Over the last few years, the identification and measurement of habitat units have become important tools for quantifying fish habitat and identifying limiting factors for fish populations (e.g., Bisson et al., 1982; Hankin and Reeves, 1988). Observations of change in individual habitat units, the relative abundance of different units, or the

sequence of units, represent a quite different use of the methodology and a class of monitoring techniques that currently are under extensive investigation.

Physical parameters used to separate habitat units include channel slope, depth, bed material, roughness, and flow velocity. Since each of these parameters is continuously variable rather than discrete, the designation of habitat boundaries is somewhat arbitrary. Different studies have used different classification criteria, although most typically distinguish about five major habitat types (e.g., Platts et al., 1983; Ralph, 1989). Many researchers subdivide the two basic categories of pools and riffles into different subtypes (e.g., plunge pools, lateral scour pools, backwater pools, low-gradient riffles, rapids, and cascades).

Both the size and the classification of individual habitat units are flow dependent; that is, they increase or decrease in area and volume, and even the classification of individual habitat units may change with a change in discharge. The effect of a change in flow is not consistent among habitat types: for example, as flow increases, dammed pools become larger, while low gradient riffles and scour pools may become glides. Habitat unit surveys must be carried out at similar flow conditions in order to be comparable (Platts et al., 1983).

Data on the frequency and size of individual habitat units can be used to determine the relative proportion of each habitat type within a stream reach. Ratios or indices of habitat abundance can then be constructed. The pool-riffle ratio is by far the most common of these, and this has been used extensively by fish habitat managers to assess the need for habitat rehabilitation.

Relation to Designated Uses

Habitat composition provides the basis for a relatively direct link between the physical processes governing stream morphology and the suitability of the stream for fish reproduction and growth. The spatial distribution and abundance of different habitat units are critical to the relative success of different fish species. Streams that have a high proportion of riffles with a gravel substrate, for example, probably will have few large obstructions and an abundance of coarse sediment. The relative paucity of rearing habitat in such streams is likely to limit the population of some fish species or life stages while perhaps favoring others. Models used to predict habitat value require data on the frequency and abundance of different habitat types (e.g., Bovee, 1982). Any change in the flow regime or in the distribution of habitat units can be expected to alter the suitability of the stream for different fish species and the overall fish community dynamics.

An inventory of habitat types can provide an overall summary of both channel morphology and habitat complexity. Repeated surveys can show whether a shift in the relative proportions of habitat types has occurred, and any change can be related to both cause (i.e., the physical processes causing the change) and effect (change in value of fisheries resources).

Response to Management Activities

The relatively recent development of habitat unit surveys means that very few data are available on how the overall distribution of habitat types changes in response to management activities. However, existing knowledge of sediment transport and other stream processes can be used to predict how particular habitat units might change given a specific management impact (e.g., Lisle, 1982; Sidle, 1988). For example, in all but the steepest streams an increase in coarse sediment would be expected to reduce the area and volume of pools and increase the percentage of stream area occupied by riffles (Section 4.3). Similarly, a reduction in large woody debris removes important pool-forming elements, and this should increase the area of riffles and decrease the number and size of pools (Section 5.7). In stable reaches with abundant fine sediment, an increase in the size of peak flows may lead to an increase in scour pools or glides, and a corresponding decrease in riffles.

Table 6 in Part I qualitatively ranks the sensitivity of habitat units to each of the other monitoring parameters discussed in these *Guidelines*. By combining this information with Table 3 in Part I (sensitivity of the parameters to particular management activities), one can determine which management activities are most likely to affect habitat units. Predicting the precise type, rate, and magnitude of change will depend on the stream reach being evaluated and the local knowledge of stream processes.

Measurement Concepts

Quantitative habitat data are obtained by identifying and measuring individual habitat units within a designated stream reach (e.g., Bovee, 1982). The typical procedure is for a two-person crew to walk a stream channel, with one person measuring individual habitat units while the other person records the data. Hankin (1984) recommended that stratified sampling be utilized to increase efficiency and reduce error. This concept has led to the procedure of visually estimating the area of each habitat unit, and measuring a systematic sample of each habitat type to develop a correction factor for the visual estimates (Hankin and Reeves, 1988). This procedure has been widely adopted in the Pacific Northwest, as it allows an experienced two-person crew to inventory approximately 1-3 miles of stream channel per day (G. Reeves, U.S.F.S. Pac. Northw. Res. Sta., Corvallis, pers. comm.). Generally the data are used only to generate summary statistics, and changes in individual channel units, or in the sequence of units, are not evaluated.

Use of stratified sampling does not resolve the basic problem of how to classify habitat types. Some investigators identify only riffles and pools because other habitat types, such as glides, runs, and pocket waters, cannot be systematically identified (Platts et al., 1983). Others (Bisson et al., 1982) have employed habitat classification systems

that include subsets of the major habitat types (riffles and pools) because this more detailed classification system may provide more insight into the suitability of the stream for different fish species.

As noted earlier, habitat composition varies with discharge, and this must be considered when undertaking stream surveys. Observers should be given similar training in order to ensure consistency. Repetitive surveys should be conducted by the same people wherever possible in order to eliminate any bias between surveyors. If specific habitat units are being monitored, particular care must be given to defining the boundaries between adjacent habitat units, as demarcation errors will reduce the accuracy of the procedures and hence the ability to detect change (Platts et al., 1983).

At this point there are little or no data to indicate whether it is best to monitor individual habitat units or to utilize summary statistics for a stream reach. Some researchers posit that changes in the sequence of habitat units may be one of the most sensitive and revealing monitoring techniques that can be derived from habitat unit surveys.

Standards

Currently there are no regulations or standards for habitat composition. In some National Forests pool-riffle ratios are being monitored, and a decline in this ratio is considered an adverse management effect. Often a pool-riffle ratio of 1:1 is considered optimal, but the limited literature suggests that this is highly variable among streams and fish species, and should not be utilized as a standard (Platts et al., 1983).

Current Uses

An inventory of habitat units usually is conducted to assess the suitability of the stream for fishery resources. Unfortunately, "ideal" conditions are difficult to define and are likely to vary widely according to the fish species of interest, the flow regime, and other environmental factors. Hence we may be able to identify stream reaches that have clearly been impacted by land management activities and offer poor quality habitat for salmonids, but it may not be possible to clearly rank streams classified as "acceptable." Thus one benefit of conducting habitat surveys will be a better understanding of the existing variability of habitat units among streams. To the extent that fish census data are available, and other factors such as fishing pressure can be accounted for, it should be possible to better define "ideal" habitat conditions.

Use of habitat units for monitoring environmental change has not been extensively tested because of the paucity of long-term data. Extensive stream surveys that estimate or measure each habitat unit only recently have been initiated in Washington, Oregon, and Idaho by agencies such as the U.S. Forest Service. Much of the data have not yet been analyzed, but the results are expected to document a large amount of variability in undisturbed streams. Subsequent surveys will be needed to determine what level of change is acceptable and how to distinguish changes due to land management activities from changes due to natural causes. A few repeat surveys have at least indicated that survey data are consistent (S. Ralph, Univ. of Washington; D. Bates, Gifford Pinchot Natl. Forest; and G. Luchetti, King County, WA, pers. comm.).

Assessment

Habitat unit surveys provide a useful, quantitative characterization of stream channels. At this point, however, our ability to classify and measure habitat units probably exceeds our capability to interpret the results. This should change as comparative data become available and the results of individual surveys are linked to land management activities. As with other geomorphic parameters, it may prove difficult to separate land use effects from the effects of natural events.

Habitat unit surveys may be relatively insensitive to land use practices. A small amount of sediment, for example, might significantly alter the bed material (Section 5.6) or residual pool depth (Section 5.3), but might not alter the size of, or ratios among, different habitat units. We should expect that different habitat units will exhibit differences both in their sensitivity to change, and in their recovery rate once change does occur. More experience is needed to determine if it is better, for example, to directly monitor pool parameters (Section 5.3) or large woody debris (Section 5.7) rather than habitat units. In view of this uncertainty, current efforts to conduct large-scale habitat unit surveys must be viewed with some concern.

In summary, habitat unit surveys are important to improve our knowledge of the relationship between aquatic life, fish production, and stream channel morphology. By then linking habitat data to land use activities and climatic events, we can better define optimal conditions and susceptibility to change. At present, however, we do not have the experience or data to fully assess the potential of habitat unit surveys as a monitoring technique.

5.6 Bed Material

5.6.1 Particle-Size Distribution

Definition

The composition of the material along the stream bed is a very important feature of stream channels. The most common method to characterize the bed material is to classify it by particle size. By taking a sufficiently large sample, one can construct a plot of particle size versus frequency in percent.

Different points in the particle-size distribution are used to provide a simple characterization of the bed material. Common variables include the median particle size (d_{50}) and d_{84}, which is the particle diameter equal to or larger than 84% of the particles (clasts) on the channel bottom. The d_{84} and d_{16} are used to describe the variability of the particle-size distribution around the mean because they are each one standard deviation away from the mean when the data are transformed onto a logarithmic scale.

Another approach to evaluating the bed material is simply to estimate or measure the percent of the bed surface covered by fine particles. The size limit for fine particles will vary by location and purpose of the monitoring, but usually ranges between 2 and 8 mm in diameter. This approach implicitly assumes that fine sediment is of primary concern, and it is not necessary to determine the size distribution of the coarser bed materials.

Chapman and McLeod (1987) conclude that the fredle index shows some promise as a measure of gravel suitability for salmonid spawning in the Northern Rockies. The fredle index is defined as d_g/s_g, where d_g is the geometric mean particle size, and s_g is the geometric standard deviation (Lotspeich and Everest, 1981).

Relation to Designated Uses

The particle size of the bed material directly affects the flow resistance in the channel, the stability of the bed, and the amount of aquatic habitat (Beschta and Platts, 1986). Because the flow resistance is one part of the overall energy loss in streams, the mean particle size can be related to the other factors that control energy loss in streams such as the stream gradient (Hack, 1957) and the sinuosity.

Although a direct relationship exists between the size of the bed material and the stability of the bed, other factors such as the slope, depth, local turbulence, and bank characteristics will affect whether a particular particle will be moved. The frequency of bedload transport is of critical importance for fish spawning and the other organisms utilizing the stream bottom for cover, foraging, or as a substrate.

The size of the bed material also controls the amount and type of habitat for small fish and invertebrates. If the bed is composed solely of fine materials, the spaces between particles are too small for many organisms. Coarser materials provide a variety of small niches important for small fish—especially juvenile salmonids—and benthic invertebrates. Coarser materials also have more interflow through the bed, effectively expanding the suitable habitat for benthic invertebrates and other organisms down into the stream bed, and facilitating salmonid reproduction. Platts et al. (1979) found a close relationship between geometric mean particle size and gravel permeability. Hence a decrease in the median particle size of bed material will decrease the permeability of the bed material, and this will tend to decrease intergravel dissolved oxygen (DO) concentrations. Even a small decline in intergravel DO can severely affect the survival of salmonid eggs, alevins, and invertebrates (Section 2.4).

Effects of Management Activities

One of the most common and probably the most damaging effect of forest management activities is to decrease the median bed material particle size. Forest harvest, road building and maintenance, and placer mining all tend to increase erosion and sediment delivery rates (Swanson et al., 1987). Most of the material reaching the stream channel as a result of human activities will be sand-sized or smaller. The deposition of this material in the stream channel then has a series of adverse effects (Chapter 4; Everest et al., 1987).

There is some evidence that an increased deposition of fine materials may be partially self-perpetuating. In some cases the onset of bedload transport is delayed when the interstitial spaces are filled with fine sediment (Reid et al., 1985). A reduced frequency of bedload transport then provides more opportunity for the deposition of fine particles and fewer opportunities for fines to be washed out during high flows (Beschta and Jackson, 1979).

Measurement Concepts

The characterization of bed material has been the subject of considerable study. Pebble counts are used to develop a particle size distribution for the bed surface material, while bulk samplers are used to determine the particle size distribution in the surface or subsurface. The selection of a measurement technique depends on the time and equipment available, as well as on the objectives of the sampling.

Pebble counts are a systematic method of sampling the material on the surface of the stream bed (Wolman, 1954). Typically a grid or transect is established, and the sizes of 100 or more particles are tabulated to establish a frequency distribution. Since each sampled particle represents a portion of the bed surface, the frequency distribution represents the percent of the stream bed covered by particles of a certain size, and not the percent by volume or weight. Particles smaller than 2-4 mm are difficult to measure in the field and may be classified only as fines (Wolman, 1954). Other studies estimate the size of fine particles by feel or comparison to reference samples. Pebble counts are simple and rapid, but there may be some bias against selecting very small or very large particles.

A second approach to determining the particle-size distribution of the bed material is by obtaining and sieving bulk samples. A McNeil sampler is the most common means to obtain a bulk sample. The McNeil sampler is a metal, tube-shaped device that is driven into the streambed to the desired sampling depth. Coarse material within the sample tube is extracted by hand. By capping the tube when extracting the corer most of the fine sediments are retained (McNeil and Ahnell, 1964; Platts et al., 1983). The other

major technique to obtain a bulk sample is to freeze a sample of the bed material using liquid CO_2 or liquid nitrogen. The frozen sample is then thawed and sieved in order to obtain the particle size distribution. One major advantage of frozen cores is that they retain the vertical structure in the sample, thereby permitting comparisons between particle-size distributions at different depths (Section 5.6.3). Platts et al. (1983) discuss both these techniques in detail and conclude that (1) neither the McNeil sampler nor the freeze core technique is adequate when substrate particles larger than about 25 cm are present, and (2) neither takes a completely representative sample.

One difficulty with evaluating the extensive literature on bed material particle size is the variation in the systems used to classify particle sizes. Some investigators have used many size classes, while others have used as few as six size classes (Platts et al., 1983; Chapman and McLeod, 1987). Each size class can be associated with a specific term (e.g., sand, gravel, cobbles, boulders), but these terms are not necessarily consistent (Platts et al., 1983). The most common classification system in the U.S. is presented in Table 9. A classification commonly used in the scientific literature is the phi index, where phi = $-\log_2 d$, with d being the particle diameter in mm. Use of the phi index normalizes the particle-size distributions so they can be analyzed using parametric statistics and plotted directly on arithmetic graph paper (Wolman, 1954).

The selection of the sampling technique should be determined by the objectives of the sampling. Characterization of the bed material can be done most easily by using Wolman pebble counts or by measuring the percent of the bed surface covered by fines. McNeil core samples and freeze cores both are useful in assessing the suitability of the substrate as spawning gravel. Freeze cores can be used to determine the variation in the particle-size distribution with depth. Comparisons between the surface and subsurface samples may indicate a change in the sediment load (Dietrich et al., 1989; Section 5.6.3).

Table 9. Classification of bed material by particle size (adapted from Platts et al. 1983).

Class name	Size range		
	Millimeters	Inches	ϕ*
Very large boulders	4,096 - 2,048	16 - 80	-12 - (-11)
Large boulders	2,048 - 1,024	80 - 40	-11 - (-10)
Medium boulders	1,024 - 512	40 - 20	-10 - (-9)
Small boulders	512 - 256	20 - 10	-9 - (-8)
Large cobbles	256 - 128	10 - 5	-8 - (-7)
Small cobbles	128 - 64	5 - 2.5	-7 - (-6)
Very coarse gravel	64 - 32	2.5 - 1.3	-6 - (-5)
Coarse gravel	32 - 16	1.3 - 0.6	-5 - (-4)
Medium gravel	16 - 8	0.6 - 0.3	-4 - (-3)
Fine gravel	8 - 4	0.3 - 0.16	-3 - (-2)
Very fine gravel	4 - 2	0.16 - 0.08	-2 - (-1)
Very coarse sand	2.0 - 1.0	0.08 - 0.04	-1 - (0)
Coarse sand	1.0 - 0.5	0.04 - 0.02	0 - 1
Medium sand	0.50 - 0.25	0.02 - 0.01	1 - 2
Fine sand	0.250 - 0.125	0.01 - 0.005	2 - 3
Very fine sand	0.125 - 0.062	0.005 - 0.0025	3 - 4
Coarse silt	0.062 - 0.031	-	4 - 5
Medium silt	0.031 - 0.016	-	5 - 6
Fine silt	0.016 - 0.008	-	6 - 7
Very fine silt	0.008 - 0.004	-	7 - 8
Coarse clay	0.004 - 0.0020	-	8 - 9
Medium clay	0.0020 - 0.0010	-	9 - 10
Fine clay	0.0010 - 0.0005	-	10 - 11
Very fine clay	0.0005 - 0.00024	-	11 - 12

*phi.

Standards

Currently there are no existing or proposed standards for bed material particle size. The state of Idaho has been considering the use of percent of fines on the bed surface as a criterion, but this was rejected because the percent of fines on the bed surface could not be directly linked to specific designated uses of water (Harvey, 1988).

Current Uses

Bed material particle size has been used extensively in research, stream classification, stream inventories, and stream monitoring. Some monitoring projects have successfully used visual estimates or photographic comparisons to estimate particle size or percent fines (e.g., Megahan et al., 1980). Generally visual techniques are less sensitive and less reliable than the more systematic and quantitative sampling methods (Chapman and McLeod, 1987).

Both pebble counts and McNeil core samples have been used extensively by the U.S. Forest Service to inventory and monitor stream condition, but the resulting data remain largely unpublished. Long-term studies on the effectiveness of bed material particle size as a monitoring technique are surprisingly scarce, although a number of studies have investigated the effect of logging on bed material particle size with varying results (e.g., Platts and Megahan, 1975; Megahan et al., 1980; Sheridan et al., 1984; Scrivener, 1988). Probably much of this variation in results is due to the different geologies and stream characteristics. Bed material particle size is probably less appropriate as a monitoring technique in areas where clays and silts predominate, or in very steep gradient streams.

Assessment

Bed material particle size may have considerable promise for monitoring purposes as it appears to be relatively sensitive to changing sediment loads (e.g., Megahan et al., 1980; Platts et al., 1989). Additional effort is needed to more precisely define the parameter(s) to be monitored, to strengthen the link between bed surface particle size and various designated uses, and to determine the environments in which a bed material parameter is most useful.

The selection of a bed material monitoring parameter should consider whether a complete particle size distribution is needed, or whether a single number, such as the d_{50} or percent fines, will suffice. Chapman and McLeod (1987) suggest that geometric mean particle size and percent of the bed surface covered by fines should both be used to define habitat quality.

Sampling locations also need to be clearly defined. An ideal sampling location has a high sensitivity to management impacts and minimal response to natural events. Since these two criteria are likely to be in conflict, detailed studies are needed to determine the most appropriate sampling location(s) within a stream channel. Some studies suggest that percent fines should be evaluated within the egg pockets of salmonid fishes, as these have the lowest variability and the most direct link to a designated use (spawning success of coldwater fishes) (Chapman and McLeod, 1987).

Chapman and McLeod (1987) reviewed the linkages between bed material particle size and quality of fish habitat. Large amounts of fine sediment clearly are detrimental to salmonid reproduction and rearing, but quantitative relationships at lower levels of fine sediment are more difficult to establish (Everest et al., 1987). These quantitative relationships also are likely to vary among ecoregions, suggesting a need for varying standards or criteria.

In some areas, bed material particle size may not be a useful monitoring parameter. Steep headwater streams, streams with a clay substrate, and low-gradient rivers all may exhibit little change in their bed material particle-size distribution despite a changing sediment load.

The timing of sampling also may affect the results. At high flows the finer particles tend to be flushed or washed from a coarse-bedded stream. Hence sampling immediately after a high flow may indicate a coarser streambed surface than sampling after a relatively quiescent period (Adams and Beschta, 1980).

These constraints in using bed material particle size for monitoring may be alleviated by combining particle size data with other channel parameters. Monitoring of bed material particle size, for example, might be done on selected cross-sections or in selected pools and riffles within a thalweg profile. This would permit changes in bed material to be more directly linked to deposition or scour, as well as to changes in the quality and amount of fish habitat. Monitoring bed material particle size within cross-sections or a thalweg profile also simplifies the problem of identifying sampling sites. In general, a combination of techniques will facilitate cross-verification and our understanding of stream response to management activities.

5.6.2 EMBEDDEDNESS

Definition

In streams with a large amount of fine sediment, the coarser particles tend to become surrounded or partially buried by the fine sediment. As shown in Figure 8A, embeddedness quantitatively measures the extent to which larger particles are embedded or buried by fine sediment. The measure was first used to quantify stream sedimentation in the 1970s and early 1980s (Klamt, 1976; Kelly and Dettman, 1980). Since then the method has undergone a series of modifications and has been used as an indicator of the quality of over-wintering juvenile salmonid habitat (Munther and Frank, 1986; Burns and Edwards, 1987; Torquemada and Platts, 1988; Potyondy, 1988). The method and its application continue to be improved and standardized by researchers in Idaho (Skille and King, 1989) and Montana (Kramer, 1989).

Currently variation exists in the suggested minimum and maximum size of rocks to be measured and in the specific feature being measured. Most researchers define the technique as cobble embeddedness, even though measurements typically are made on all rocks with a primary axis between 4.5 cm (very coarse gravel) and 30 cm (small boulders). Torquemada and Platts (1988) modified the method to measure rocks as small as 1.0 cm, and the inclusion of these smaller particles led them to use the term embeddedness rather than cobble embeddedness.

The difficulty in measuring cobble embeddedness and the high variability of individual measurements have stimulated research into a series of related measurements. One alternative is to measure the height of the rocks above the bed surface, and this is termed "total free space" (Fig. 8B). Conceptually this is similar to bed roughness, and it is an indicator of the area protected from the current. Such areas are important fish rearing and macroinvertebrate habitat. This measurement also has been termed "living space" by Skille and King (1989) and "interstitial space" by Kramer (1989).

To reduce the variability associated with measurements from individual particles, Kramer (1989) suggested that the total free space from all particles within a specified sample area (typically a 60-cm diameter circle) be summed and then divided by the area sampled. This was termed the "interstitial space index" (ISI), where

$$ISI = \Sigma D_f / Area.$$

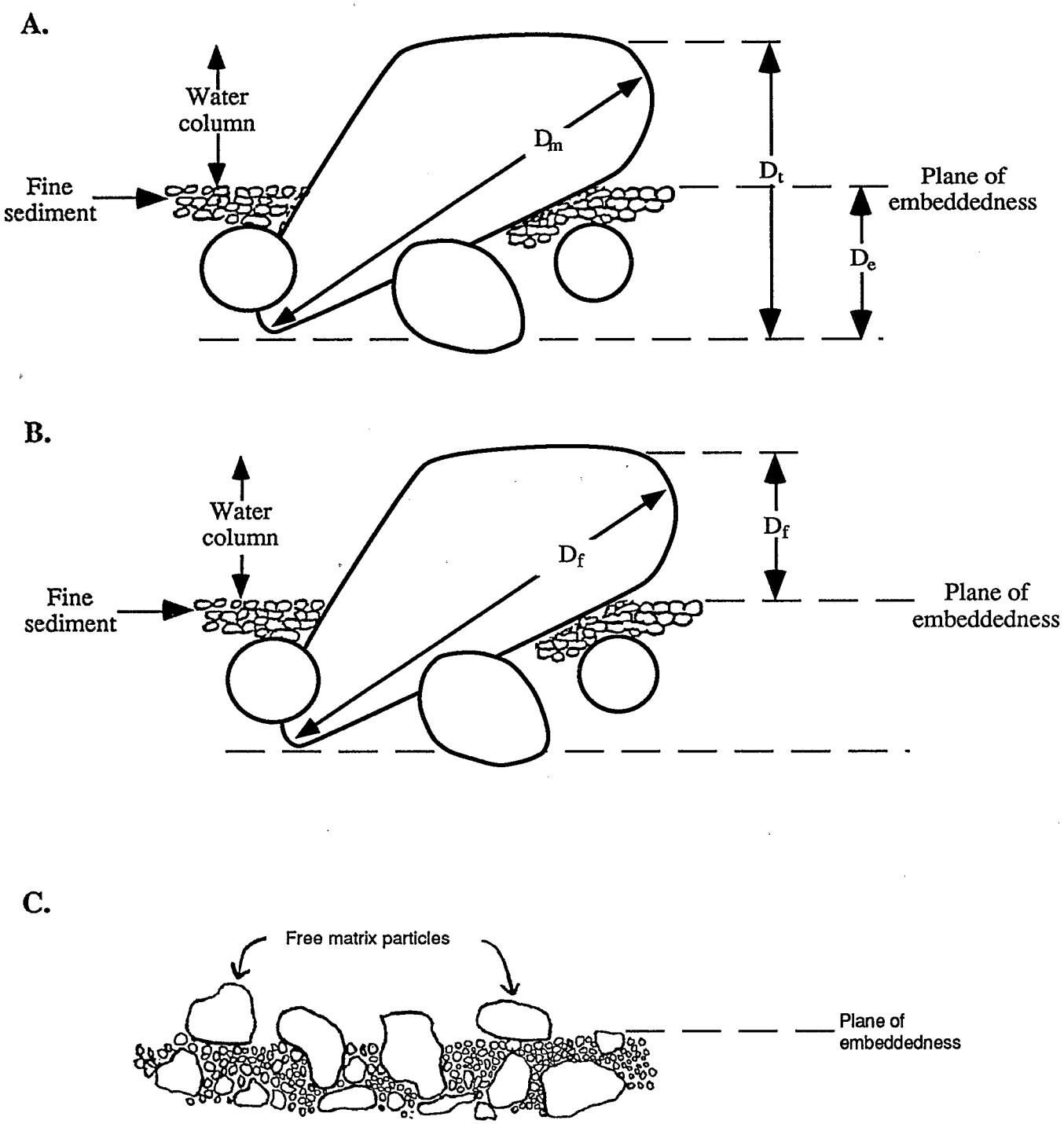

Figure 8. Schematic representation of the three main embeddedness measurements—embeddedness, free space, and free matrix particles. D_m represents the length of the primary axis. A. Embeddedness for a single particle is equal to D_e/D_t. B. Free space for a single particle is equal to D_f (note: $D_f = D_t - D_e$). C. Free matrix particles. (Adapted from Burns and Edwards, 1985.)

An average ISI can be determined for each sampled stream reach. ISI appears to be more sensitive to change than cobble embeddedness, and it also is more directly related to the designated use of streams for fisheries.

A third embeddedness measure (Fig. 8C) is the percent of free matrix particles. Free matrix particles are defined as those rocks (typically 4.5-30 cm along the primary axis) having zero embeddedness (Fig. 8C; Burns and Edwards, 1985). Percent free matrix is calculated by dividing the number of free matrix particles by the total number of similarly sized particles within the sampled area. Percent free matrix particles correlates closely with percent embeddedness (Burns and Edwards, 1985; Torquemada and Platts, 1988; Munther and Frank, 1986; Potyondy, 1988).

Relation to Designated Uses

Cobble embeddedness has both biological and physical significance. Biologically, areas with a high embeddedness have very little space for invertebrates or juvenile fish to hide or seek protection from the current. The accumulation of fines also fills in the spaces between larger particles, and this limits the interstitial habitat. Similarly, the reduction in surface area associated with increasing embeddedness (decreasing total free space) limits the attachment area for periphyton.

Chapman and McLeod's (1987) review noted lower aquatic insect densities when embeddedness exceeded 65-75%. Salmonid density also declined with an embeddedness of 50% or more. It was inferred that an increase in embeddedness would reduce winter habitat, with the precise relationship varying according to the fish species and fish population density.

The physical effects of embeddedness are similar to the effects of a decrease in bed material particle size discussed in Section 5.6.1. Increasing embeddedness decreases channel roughness, and the resulting reduced bed friction losses will have repercussions on the stream hydraulics and overall channel morphology. Total free space is closely related to bed roughness and may be proportional to Manning's "n."

The fine particles associated with increasing embeddedness adversely affect gravel permeability and intergravel dissolved oxygen. Chapman and McLeod (1987) note that an abundance of fine particles in the interstices of the bed may delay the onset of bed movement during high flows, and this in turn could facilitate the accumulation of fine particles.

Response to Management Activities

The use of cobble embeddedness for water quality monitoring presumes that increasing embeddedness reflects an increased input of fine sediments to the stream channel. Measurements of embeddedness on 19 tributaries to the South Fork Salmon River in Idaho indicated that streams in heavily roaded and logged watersheds had a significantly higher cobble embeddedness than undisturbed watersheds (Burns and Edwards, 1985). No differences were found between undisturbed and partially disturbed watersheds.

In 1986 embeddedness was sampled on 120 streams in the Boise National Forest (Potyondy, 1988). No statistically significant differences in mean embeddedness were found between developed and partially developed watersheds. Nevertheless, the study concluded that there is a relationship between mean embeddedness and sediment-producing activities, but both natural and management-induced factors are important in determining embeddedness levels.

Studies on the Payette National Forest in Idaho compared embeddedness levels in watersheds with different degrees of mining activities (Burns and Ries, 1989). The study concluded that at least 5 consecutive years of data are needed to evaluate trends in embeddedness.

Measurement Concepts

The basic procedure for measuring embeddedness is to select a particle, remove it from the streambed while retaining its spatial orientation, and then measure both its total height (D_t) and embedded height (D_e) perpendicular to the streambed surface (Fig. 8A). Percent embeddedness is calculated for each particle until at least 100 particles are measured. Individual embeddedness values are averaged to yield a mean embeddedness value.

The technique as modified by Skille and King (1989) uses 60-cm diameter hoops as the basic sample units. The total height (D_t) and embedded height (D_e) are measured for each particle which meets the specified size criterion. The individual values of D_t and D_e from each hoop are summed, and a percent cobble embeddedness (PCE) for each hoop is calculated from the formula:

$$PCE = \Sigma D_e / \Sigma D_t$$

An average of the PCE values from all the hoop samples yields the percent cobble embeddedness for the sampled reach.

The number of hoops needed to characterize a site depends on the variability among hoop samples and the desired level of precision. A general rule is that one reach requires approximately 20 hoops (approximately 500-700 particles) and may require up to 1 full day for a two-person field crew to complete.

The use of hoops rather than individual particles as the basic sampling unit substantially increases the number of particles that must be measured, but reduces the variability among sample units. This makes it easier to detect change (Part I, Section 3.4.2) and results in an embeddedness value that more closely represents the condition of the stream reach. The earlier technique of using individual particles as the sample unit may be more applicable within one habitat type where the variability is likely to be lower.

In developing a monitoring plan using embeddedness, the objectives will dictate whether hoops or individual particles should be the sample unit. To characterize a stream reach with different habitat types, Skille and King (1989) suggest three randomly spaced hoop samples along cross-sectional transects placed two stream widths apart.

The embeddedness value for a randomly placed hoop should be adjusted if the hoop incorporates a substantial area of fine sediments with no exposed rocks (Torquemada and Platts, 1988). Failure to correct for the area occupied by fines will cause embeddedness to be underestimated. The corrected value is known as the weighted embeddedness, and it is defined as:

$$WE = \frac{HA \times 100 + (1-HA)E}{100}$$

where WE = percent weighted embeddedness,
HA = percent of hoop area occupied by fines, and
E = percent embeddedness.

Skille and King (1989) suggest that the weighted value should be used if more than 10% of the surface area within the hoop is occupied by fine sediment.

The size of the particles and the diameter of the hoop can be adjusted according to the type of stream. Most recent studies have used hoops 60 cm in diameter and measured all particles with a primary axis of 4.5-30.0 cm. Fines are usually defined as particles less than 6.4 mm (0.25 inches) in diameter. These particle and hoop sizes are believed appropriate for streams up to 20 feet wide and with a gradient of up to 3% (Skille and King, 1989). Torquemada and Platts (1988) modified the method for use in smaller streams by reducing the hoop diameter to 30 cm and decreasing the minimum rock size to 1.0 cm.

The time required to evaluate embeddedness can be substantially reduced by measuring the height of free matrix particles and counting the remaining embedded particles. Since the relationship between percent cobble embeddedness and percent free matrix particles may vary according to stream order, geology, climate, etc., inferences about percent embeddedness cannot be made from free matrix data until the interrelationship has been defined for that site.

If the monitoring objective is to evaluate changes in the deposition of fine sediments, the interstitial space index (ISI) may be the preferred embeddedness parameter. Both the ISI and percent embeddedness can be calculated from one set of field measurements.

Standards

The State of Idaho Water Quality Bureau currently is proposing a cobble embeddedness criterion. This specifies that cobble embeddedness in fry overwintering habitat should not exceed natural baseline levels at the 95% confidence level. Baseline levels of cobble embeddedness are to be determined in similar watersheds that are unaffected by nonpoint sediment sources (Harvey, 1989).

Current Uses

Ongoing, unpublished studies by federal and state agencies are measuring embeddedness as one means to assess the effects of land management activities on streams. Use of the revised measurement techniques and more intensive sampling should allow a better evaluation of the usefulness of embeddedness to monitor the effects of management activities.

Currently embeddedness is being measured in a number of National Forests, particularly in Idaho and Montana. Embeddedness also is part of the Forest Practices BMP Effectiveness Monitoring Program in Idaho. In Washington four classes of embeddedness are being visually estimated in the Timber-Fish-Wildlife stream survey program. These field applications will help evaluate the methodology for measuring embeddedness and determine its usefulness for assessing the effects of past and present management activities.

Assessment

Current research and monitoring efforts should help clarify the links between embeddedness, other characteristics of the stream channel, and fisheries. Measurement of one or more embeddedness parameters (percent cobble embeddedness, total free space, or percent free matrix particles) probably will prove useful only in certain environments and stream types. Most of the work on embeddedness has been conducted in granitic basins in Idaho, and embeddedness may not be as appropriate in basins where most of the anthropogenically induced sediment load is comprised of silts and clays. Similarly, embeddedness may not be a useful monitoring parameter in high-energy, steep gradient channels where deposition of fine particles is unlikely. Low gradient downstream reaches may lack the coarse particles needed to measure embeddedness.

The strong interest in embeddedness as a monitoring parameter is due to the recognition that sediment often is the most important pollutant from forest management activities in the Pacific Northwest and Alaska. Hence there is a great need for reliable methods to evaluate sediment inputs and the resultant effects on the designated uses of the water. Embeddedness has shown promise, but the immediate need for a monitoring technique has resulted in widespread use and adaptation before cobble embeddedness could be adequately field-tested and validated. Users should be aware that the various embeddedness techniques are likely to undergo further changes and improvements, and this could severely limit the comparability of data collected over time.

5.6.3 SURFACE VS. SUBSURFACE PARTICLE SIZE DISTRIBUTIONS

Definition

The bed material in alluvial stream channels consists of mixed grain sizes. Often the surface of the bed is coarser than the underlying material. This armoring or pavement has been attributed to a settling of the smaller particles down into the bed during active transport (Parker and Klingeman, 1982), and selective transport of finer particles when the larger particles are immobile (Sutherland, 1987). Surface coarsening has been observed downstream of dams when bedload was eliminated (Shen and Lu, 1983; Bradley and Smith, 1984).

An alternative to this "equal mobility" explanation for the armoring of gravel-bedded streams and rivers is that the armoring is a result of the sediment supply being less than the sediment transport capacity (e.g., Kinerson and Dietrich, 1989). If one assumes that the subsurface particle size distribution is similar to the particle size distribution of the bedload (e.g., Parker et al., 1982) and that the banks are relatively resistant to erosion, then the difference between the surface and subsurface particle size distribution should be quantitatively linked to the sediment supply (Dietrich et al., 1989). A dimensionless ratio, q*, has been defined as the estimated bedload transport rate for the median grain size on the bed surface divided by the estimated bedload transport rate for the median grain size of the subsurface material (Dietrich et al., 1989).

Under this hypothesis streams with a high sediment load and no surface coarsening should have a high q*, while streams with a low sediment load should have a well-developed coarse surface layer and a low q*. With an increased sediment load, streams that initially had a low q* would experience a fining of the bed surface material. With a higher q*, relatively little of an increased sediment load could be accommodated by a fining of the bed surface, and the stream would be more subject to aggradation, pool filling, and overall channel instability. An increased sediment supply also would lead to a greater proportion of the stream bed being occupied by finer materials (Kinerson, 1990).

Relation to Designated Uses

The effects of an increase in the sediment supply, and the corresponding fining of the bed surface relative to the subsurface, have been discussed in Chapter 4 and in Sections 5.6.1-5.6.2. Briefly, an increase in fine sediment will decrease the permeability of the bed material in alluvial channels, which will decrease intergravel DO (Section 2.2) and degrade spawning habitat. A predominance of fine sediment decreases macroinvertebrate biomass and diversity (Chapman and McLeod, 1987; Everest et al., 1987). Mean particle size in the bed material is inversely correlated with habitat suitability for aquatic insects and fish (Chapman and McLeod, 1987). By reducing pool depth and pool volume, sediment deposition reduces the suitability of a stream for adult fish (Section 5.3). Increasing embeddedness and surface fines reduce winter carrying capacity for salmonids in the northern Rockies (Section 5.6.2). Comprehensive reviews of the effects of sediment on aquatic organisms are presented in Chapman and McLeod (1987) and Everest et al. (1987). Scrivener (1988) summarizes the forest management-sediment-fisheries interactions for the Carnation Creek study in coastal British Columbia.

Effect of Management Activities

The impact of forest management on sediment production is discussed in Chapter 4. Swanson et al. (1987) and Everest et al. (1987) both provide excellent overviews of natural sediment production rates, the processes governing the input of sediment into streams, the impact of sediment on aquatic ecosystems, and the extent to which forest management activities are likely to increase sediment production rates. Swanson et al. (1987) conclude that mass failures are the dominant source of sediment, but the processes that deliver sediment to the stream channel are more variable. Forest management activities—particularly road building, poor road maintenance, and the combination of clearcutting and broadcast burning—usually have the greatest effect on sediment yields. Steeper basins appear to be more sensitive to management impacts, and evaluating management impacts is complicated by the random occurrence and potential impact of large storm events (Swanson et al., 1987). Everest et al. (1987) note that while the felling and bucking of trees can have minimal impact on fine sediment production and yield, roads, tractor logging, and ground-disturbing site preparation activities tend to have a much larger impact.

A long-term study in the South Fork of the Salmon River in Idaho showed that for the first 10 years after a logging moratorium was imposed the percent of fines (<4.75 mm in diameter) declined relatively rapidly in both the surface and subsurface layers (Box 3, page 19; Platts et al., 1989). This was followed by a period of less rapid decline, and from about 1981 to 1985 there was a small increase in percent fines. Surface fines were removed more rapidly than subsurface fines because they were more exposed to the shear stress imposed by the flowing water. On the other hand, once an apparent state of equilibrium had been reached, the percent of fines in the surface layer remained at approximately half the concentration found in the subsurface layer. Considerable variation was found between monitoring sites, and this was partly attributed to differences between low-gradient spawning areas and higher-gradient rearing habitats (Platts et al., 1989).

The Carnation Creek study in coastal British Columbia monitored changes in particle size distribution in the top (0-15 cm) and bottom (15-30 cm) layers of bed material over a 13-year period. Within and below the area of intense streamside logging, the accumulation and cleansing of fines was highly responsive to both the input of sediment and the occurrence of runoff events. Chronic sedimentation resulted in fines penetrating deeper into the streambed, and these deeper layers were much slower to recover because scour to these depths was much less frequent. Hence the annual rate of change in the particle-size distribution declined with increasing particle size and increasing depth. Significantly, 8 years after the intensive logging treatment the changes in gravel composition were still accelerating, and fine particles were still accumulating in the deeper layers (Scrivener, 1988).

Measurement Concepts

Different techniques can be used to sample the surface and subsurface bed material (Section 5.6.1). Particle-size distributions for the bed surface can be obtained by pebble counts, McNeil samplers, or freeze cores. Pebble counts allow rapid determination of the particle-size distribution of the surface layer, but this method cannot be used for the subsurface layers. McNeil samplers do not allow separation of material by depth, and this limits their use to situations where separate samples can be taken from the surface and subsurface layers. Freeze cores sample both the surface and subsurface layers, and they preserve the spatial structure of the sample. However, freeze cores are difficult to obtain in the field, and—like McNeil samplers—they are limited in terms of the maximum particle size that can be sampled (Platts et al., 1983; Section 5.6.1).

Data on the particle-size distribution in the surface and subsurface layer can be analyzed in several different ways. The simplest method is to compare the median (d_{50}) particle size of the surface and subsurface materials. Since quite different particle-size distributions can have a similar d_{50} (Platts et al., 1983), comparisons generally should incorporate some measure of the variation in the particle-size distribution, such as the d_{84} and the d_{16} (where d is diameter, and the number is the percent of particles that are smaller than the specified percentage). In cases where the particle size distribution of the surface and subsurface layers is known, one should consider developing a statistical measure of the differences between the two distributions.

Standards

No standards for the relationship between surface and subsurface particle-size distributions have been established or proposed.

Current Uses

Values of q* have been determined for a series of flume experiments (Dietrich et al., 1989) and a number of streams in California with a widely varying sediment supply (Kinerson and Dietrich, 1989). The data collected to date shows that rivers and streams with a high sediment supply generally lack a coarse surface layer and have a q* close to 1.0. Considerable local variation occurred within- stream reaches. In sediment-rich streams, for example, areas with an armor layer and a low q* could be found immediately downstream of debris jams and other obstructions which functioned as sediment traps (Kinerson and Dietrich, 1989). Chapman and McLeod (1987) also noted large differences in particle-size distributions between salmonid egg pockets and immediately adjacent areas. This instream variability should be minimized by selecting relatively straight, featureless reaches with little form roughness.

Some studies on the infiltration of fine sediment into gravel layers or redds suggest that further work is needed before the difference in particle-size distribution or q* can be adopted as a monitoring technique. Beschta and Jackson (1979) showed that the relative size differences between coarse and fine bed material can greatly affect the behavior of fine particles. When sands with a median particle size of 0.5 mm were added to a clean gravel bed with a median particle size of 15 mm, the sand was trapped in the interstitial spaces within the uppermost top 10 cm. Reducing the median diameter of the sand to 0.2 mm allowed the sand to filter down through the gravel and the interstitial voids were filled from the bottom of the flume upwards. At Carnation Creek the fine (sand-sized) particles intruded into the gravel a few centimeters below the depth of scour, and they were not winnowed out until a subsequent event scoured to that depth. These results suggest that monitoring the bed material particle size in the surface layer may be best for evaluating short-term changes, but a comparison of the surface and sub-surface particle size distributions provides a longer-term perspective on the amount and type of sediment load. Some of the complexities of the interactions between fine sediment and alluvial streambeds were recently reviewed by Jobson and Carey (1989).

Empirical support for the use of q* or a similar measure can be derived from field observations of salmonid redds. Chapman and McLeod (1987) cite several studies in which it was observed that a seal of fine particles formed over the clean gravels created by the spawning female. In these cases the deposition of fine sediments also may be affected by the special hydraulics associated with the redd.

Assessment

The relationships between sediment supply, sediment transport capacity, and the surface and subsurface particle-size distribution are in a state of active investigation. Both

theoretical considerations and preliminary field data suggest that differences between the surface and subsurface particle-size distributions can be used to monitor changes in sediment supply. However, characterizing the subsurface particle-size distribution is not a simple task, and this may ultimately limit the usefulness of q* or an associated measure as a monitoring technique.

The primary advantage of comparing surface and subsurface particle-size distributions is that this appears to provide an immediate assessment of the sediment transport capacity in relation to the sediment supply. The use of q* normalizes the surface conditions against the particle-size distribution and predicted bedload transport capacity of the subsurface layer. This yields a single index for evaluating current conditions and comparing different streams. However, if one is concerned solely with changes over time and has time-trend data available, the surface particle-size distribution could serve as the primary monitoring parameter.

Surface and subsurface comparisons may not be necessary if predictions could be made of the bed surface particle-size distribution for streams in the absence of management activities. In other words, if the undisturbed particle-size distribution can be predicted from channel characteristics or by comparisons to other streams, the actual bed surface particle-size distribution could be compared to the predicted distribution in order to evaluate stream condition.

In summary, a variety of studies suggest a direct relationship between an increase in sediment supply due to land use and a change in the surface particle-size distribution. In most cases, however, there already have been some adverse land use impacts, and no data are available on the predisturbance particle-size distribution of the bed surface. Under these circumstances a comparison of the surface and subsurface particle-size distributions may yield a quantitative measure of the sediment supply relative to the sediment transport capacity. The variability of such a measure within a particular stream reach, and the complexity of sediment transport in alluvial channels, mean that additional work will be needed before q* or a similar measure can be adopted as a standard for monitoring management impacts.

5.7 LARGE WOODY DEBRIS

Definition

Large pieces of wood in streams have been referred to by a variety of names including large organic debris (LOD), coarse woody debris (CWD), and large woody debris (LWD). The type and size of material included in this designation has varied according to the objectives of the person measuring the debris. Studies on the energetics of stream systems have included material as small as 2.5 cm in diameter as LWD (e.g., Harmon et al., 1986). However, studies of the effects of woody debris on channel morphology typically use a much larger minimum size for LWD—usually 10 cm in diameter and 2 m in length (Sedell et al., 1988; Bilby and Ward, 1987).

The amount of LWD in stream channels depends on a variety of factors. Stream size is an important determinant, with smaller streams usually containing more wood than larger systems (Swanson et al., 1982; Bilby and Ward, 1987). Riparian tree density is positively related to LWD amount in streams in eastern Washington (Bilby and Wasserman, 1989). Bed characteristics also have been shown to influence LWD amount, as streams with boulder or bedrock substrates typically contain only about half the LWD compared to streams with finer substrates (Bilby and Wasserman, 1989). Catastrophic events, such as major windstorms or landslides, also have a major impact on the amount and location of LWD in some stream channels (Keller and Swanson, 1979; Bisson et al., 1987).

Stream size plays a major role in determining the size of LWD in stream channels as well as the amount of LWD. Generally, the average size (diameter, length, or volume) of LWD in a stream channel increases with increasing stream size (Bilby and Ward, 1987). This increase is caused by the increased capacity of larger channels to move material downstream. Thus, in larger channels, smaller wood is selectively flushed from the system or deposited on the floodplains, leaving only the larger pieces. This causes a decrease in the amount of LWD, but an increase in average piece size. Pieces of wood with a low probability of being moved by the stream are most important in influencing channel morphology (Bilby and Ward, 1987). In general, pieces one-half the channel width in length or longer are regarded as being relatively stable (Bisson et al., 1987).

Relation to Designated Uses

LWD influences stream systems and their biota in a number of ways. Large wood has a major impact on channel form in smaller streams (Sullivan et al., 1987). The location and orientation of LWD can influence channel meandering and bank stability (Swanson and Lienkaemper, 1978; Cherry and Beschta, 1989). LWD tends to cause both a greater variability in channel width and an increase in average channel width (Keller and Swanson, 1979). LWD also forms and stabilizes gravel bars (Lisle, 1986).

LWD is often the most important structural agent forming pools in small streams. Bilby (1984b) reported that over 80% of the pools in a small stream in southwest Washington were associated with wood. Similarly, Rainville et al. (1985) found that 80% of the pools in a series of small streams in the Idaho Panhandle were wood-associated. While the relative importance of LWD in pool formation decreases with increasing channel width, wood in large rivers forms pools along the channel margins or in secondary channels, and these pools may be very important for fish populations (Bisson et al., 1987).

Another way in which wood affects channel shape is by forming waterfalls. Waterfalls form plunge pools and also influence sediment transport in streams. The greater the proportion of the drop in elevation of a stream caused by waterfalls the less efficient the system is at moving sediment downslope (Heede, 1972). The proportion of channel drop accounted for by summing the heights of LWD-caused waterfalls ranged from 30 to 80% in streams in the western Oregon Cascades (Keller and Swanson, 1979). In streams in the Oregon Coast Range, wood caused 6% of the total fall (Marston, 1982). In western Washington the proportion of elevation drop caused by LWD was found to decrease with increasing stream size. LWD accounted for >15% of the elevation drop in stream channels <10 m wide, but <5% of the elevation drop in channels 10-20 m wide.

LWD also influences sediment transport in streams by forming depositional sites. Wood was responsible for storing half the sediment in several small streams in Idaho (Megahan and Nowlin, 1976). The importance of wood in retaining sediment in small streams has been demonstrated by the release of very large amounts of material after removal or disturbance of LWD (Baker, 1979; Beschta, 1979).

LWD also can provide storage sites for leaves, twigs, and other organic material. In small streams in forested areas, this fine organic material can provide the bulk of the energy and materials entering into the aquatic food web. In the absence of LWD, much of the terrestrial organic matter entering the stream is flushed rapidly downstream with little opportunity for the biota to utilize this material (Bilby and Likens, 1980).

LWD is one of the most important sources of habitat and cover for fish populations in streams. Most of the work documenting this function of LWD has been done on salmonids in the Pacific Northwest (Sedell et al., 1984; Bisson et al., 1987; Sedell et al., 1988). Generally there appears to be a direct relationship between the amount of LWD and salmonid production; no known data indicate an upper end to this relationship (Bisson et al., 1987). One of the key functions of LWD with regard to fish production is to increase habitat complexity, and this helps ensure that cover and suitable habitat can be found over a wide range of flow and climatic conditions. LWD also may allow a finer partitioning of the available habitat. Pools formed by LWD, for example, are favored habitat by certain species and age groups of salmonids (Bisson et al., 1982). More complex wood structures, such as rootwads or small debris jams, attract more fish than single logs (Sedell et al., 1984; McMahon and Hartman, 1989). In a number of experiments, wood removal has been demonstrated to reduce fish population densities (Lestelle, 1978; Bryant, 1983; Dolloff, 1986; Elliott, 1986; Bisson et al., 1987).

Several potentially detrimental effects are associated with LWD in streams. Historically, massive wood accumulations on larger rivers impeded navigation. Most of these accumulations were removed around the turn of the century (Sedell and Luchessa, 1982). In most large rivers today, wood is found primarily along the channel margins or in off-channel areas (Bisson et al., 1987) and therefore poses little hazard to navigation.

Movement of wood during high flow events may damage structures located in or near streams. Large woody debris may also increase flood damage by partially blocking the channel during high flow events (e.g., Griggs, 1988). Often the risk of damage is exacerbated by ill-advised development in floodplain areas and changes in the hydrologic regime due to changes in land use. Most flood-routing models ignore the potential for LWD to influence water movement through a drainage system, even though this can greatly restrict channel capacity (P. Williams, P. Williams & Assoc., Ltd., San Francisco, CA, pers. comm.).

Large wood accumulations may form blockages to the passage of anadromous fishes. For many years this was perceived as a serious problem, and wood was removed from channels to prevent the formation of blockages. However, many LWD accumulations which appear to be blockages at low flows are passable at higher discharges. In addition, these blockages normally occur in steeper channels where spawning and rearing habitat for anadromous fish is limited. Historical estimates suggest only 5-20% of available anadromous fish habitat was inaccessible because of debris blockages (Sedell et al., 1984).

While LWD may contain some compounds toxic to stream biota, under most conditions leaching of these materials occurs at a very slow rate. This keeps concentrations well below toxic levels (Bisson et al., 1987). Similarly, LWD is seldom a cause for low dissolved oxygen concentrations in stream water. Wood is relatively resistant to decomposition, and LWD has a low surface area to volume ratio. Taken together, these two factors result in LWD having a low biochemical oxygen demand (Bisson et al., 1987).

Response to Management Activities

Historically, the amount of LWD in streams has been reduced as a result of several management practices. Wood in larger river systems was removed to improve navigation and reduce flooding hazards at the turn of the century (Sedell and Luchessa, 1982). Extensive clearing of wood from smaller streams was conducted through the early 1980s to reduce bank and bed scour and provide upstream passage for anadromous fish (Bilby, 1984b; Sedell et al., 1988). After channel clearing, much of the residual debris is unstable and is flushed from the stream channel, further reducing the amount of LWD (Bilby, 1984b).

The practice having the most widespread influence on LWD in Pacific Northwest streams has been the harvest of trees from riparian areas. Although the amount of LWD in streams may increase immediately after harvest owing to the introduction of logging slash, much of this material is rapidly decomposed or flushed from the system by high

flows. Harvest of the larger trees in the riparian zone removes the primary source of LWD, and this results in a gradual decrease in wood over time as inchannel material decomposes or is moved downstream (Swanson and Lienkaemper, 1978; Grette, 1985; Bisson et al., 1987). Typically the average piece size also declines because of the introduction of smaller pieces of wood and the relatively small size of the LWD contributed by the second-growth riparian vegetation (Sedell et al., 1988). Recent research indicates that the decrease in LWD following removal of riparian vegetation may occur much more rapidly than previously thought (B. Bilby, Weyerhaeuser Co., pers. comm.).

The length of time needed for riparian areas to produce LWD after harvest depends upon the size of the stream. Measurable contributions of wood from second-growth riparian areas did not occur until 60 years after harvest for third-order channels on the Olympic Peninsula (Grette, 1985). Bilby and Wasserman (1989) indicate that it takes longer than 70 years for streamside vegetation to provide stable material to streams wider than 15 m in southwestern Washington. Thus larger streams are likely to be deficient in LWD for a longer period of time after timber harvest than smaller streams.

A decline in the amount and average size of LWD in streams following timber harvest leads to a reduction in waterfalls, a decrease in pool frequency and size, and a decrease in the amount of sediment and finer organic matter retained by LWD (Bilby, 1984b; B. Bilby, Weyerhaeuser Co., pers. comm.). However, in some instances an increase in pool frequency has been associated with a decrease in LWD due to the replacement of large pools by numerous small pools (McDonald and Keller, 1983).

Relatively little is known about the importance of upstream source areas for maintaining LWD in larger rivers. If upstream areas are an important source of LWD in downstream areas, any reduction in LWD in these smaller upstream channels could have important off-site impacts. However, the capacity of a system to transport wood increases in a downstream direction. This suggests that on-site recruitment from the riparian area is the most likely source of stable LWD in larger rivers.

Measurement Concepts

Platts et al. (1987) provide a recent review of techniques to measure and map LWD. Selecting a methodology depends upon the objectives of the monitoring. Measurement techniques vary widely in terms of effort required, and they range from a simple enumeration of pieces to a detailed description of the characteristics and location of each piece. More detailed descriptions might include measuring the size of each piece, mapping the associated channel characteristics, and noting the location and orientation of each piece relative to a permanent benchmark. An alternative procedure for monitoring the stability of LWD is to tag and relocate each piece on an annual or storm basis.

Various criteria have been employed to delineate those pieces of wood to be included in an LWD survey. Most surveys include only those pieces which extend below the waterline at bankfull discharge and exceed some minimum dimensions. Surveys measuring the biomass of organic material in stream systems will use a smaller minimum size than studies of LWD influences of channel morphology or fish habitat (Harmon et al., 1986).

The cost of monitoring LWD increases considerably if volume or biomass estimates are needed, as this requires at least the length and diameter of each piece. Length measurements may include the entire piece or just that portion extending below the bankfull channel. Diameter may be measured at the mid-point of the piece or by averaging the diameter at both ends. Probably the most efficient procedure to determine volume or biomass is to visually estimate the length and diameter and then correct the visual estimate by measuring a subsample of the pieces (Hankin and Reeves 1988). Biomass of LWD also can be estimated with techniques derived from inventories of forest residues (Van Wagner, 1968). This procedure inventories all LWD intersected by a series of cross-sections across the stream (Froelich et al., 1972; Lammel, 1972), and is most applicable when the minimum piece size is relatively small. Special procedures or categories may be needed for measuring debris jams, standing trees, and snags within the stream channel (Platts et al., 1987).

Information on channel features associated with LWD is sometimes collected during surveys. Data may include the following:

- type of habitat unit or channel feature
- surface area, or volume of wood-associated pools
- surface area or volume of sediment stored behind LWD
- number and heights of waterfalls; and
- volume or biomass of fine organic matter

(Bisson et al., 1982, 1987; Platts et al., 1983, 1987; Bilby and Ward, 1987). Surveys of habitat types or channel features also may include data on the presence or absence of LWD (e.g., Ralph, 1989).

Standards

Standards for LWD in streams have not been established for any state, although an attempt was made in the development of Washington's forest practice regulations to maintain wood levels at those seen in old-growth stands (Bilby and Wasserman, 1989). However, LWD amounts and characteristics vary as a function of stream size, vegetation type, and other factors, thus inhibiting the establishment of strict numerical standards.

Most Pacific Northwest states have established Best Management Practices (BMPs) to control the adverse effects

of forest management on stream channels and riparian areas. The most recent revisions of these BMPs have incorporated provisions for retaining LWD in streams and ensuring a continuing supply from the riparian area. Approaches currently in use or being considered include defining strips along the stream in which no harvest is permitted (e.g., Alaska), establishing specific numbers of trees to be left along the stream (e.g., Washington), or establishing a minimum basal area which must be retained along the stream (e.g., Oregon).

Generally, the regulations applying to larger, fish-bearing waters are more stringent than those used on smaller streams. On larger streams the disturbance of inchannel debris, or removal of standing timber from the riparian area, is generally prohibited or restricted. Thus LWD in the channel is protected, slash introduction during timber harvest is reduced, and the future source of LWD for the channel is retained.

Although some states have developed regulations to restrict forest management activities near smaller streams, frequently slash is introduced to these smaller channels during timber harvest. In cases where the amount of slash entering the channel is considered to pose a threat to downstream resources, cleaning of the channel may be required. Factors considered in deciding whether or not to remove a piece of wood from the channel include the size of the woody debris and the extent to which it is embedded in the streambank or channel.

Current Uses

Recent programs to inventory stream condition and fish habitat on forest lands usually include some measurements of LWD. Generally the LWD measurements focus on the number and size of LWD pieces and their association with various channel features. As most of these programs are of recent origin, relatively little of the resulting data have been used to develop management prescriptions (Bilby and Wasserman, 1989).

When possible, comparable surveys should be conducted on similar, unmanaged streams. For example, upstream wilderness areas can provide reference data on the natural loading, recruitment rate, and downstream transport of LWD.

Such comparisons of logged and unlogged reaches can provide insights into management impacts on LWD. However, the long residence time of LWD in streams suggests that the ultimate impact of forest harvest on amounts, characteristics, and functions of LWD may not be evident for years or decades.

Assessment

Large woody debris (LWD) performs a variety of functions critical to the maintenance of productive fish habitat in stream systems. Various management activities, including timber harvest, alter the amount and characteristics of LWD in Pacific Northwest streams; therefore, monitoring activities evaluating stream conditions on forest lands should incorporate measurements of LWD. This need to monitor LWD is increasingly recognized, but monitoring programs with a LWD component are only now being established. The types of measurements which should be taken will depend upon the objectives of the specific monitoring project, but should include, as a minimum, wood abundance and piece size.

Logging and fish habitat improvement projects are the two activities most likely to alter the amount of large woody debris in stream channels. On-site measurements of wood frequency and piece size can be a relatively sensitive indicator of management impacts. In downstream locations changes in the LWD size and frequency usually occur more slowly and may not be easily detectable.

The long time required for a tree to mature and enter into the stream channel suggests that one should monitor the vegetation in the riparian zone and plan for future recruitment (Section 5.2). Hence long-term monitoring of large woody debris in stream channels is needed to fully assess the adequacy of present practices, whereas a simple inventory may suffice for evaluating conditions with regard to fish habitat, channel morphology, and sediment storage.

The extensive changes in forest practice regulations over the last twenty years means that long-term trends in LWD must be evaluated in the context of the regulations in force at the time of the management activity. Hence the data from long-term monitoring projects may not be directly applicable to current practices, but they can provide some guidance to the formulation of future regulations.

5.8 Bank Stability

Definition

Stream and river banks control limit the lateral movement of water. Typically the bank areas can be identified by a change in substrate and a break in slope between the channel bottom and the stream banks. In many streams the slope of the bank exceeds 45° (Platts et al., 1987).

Bank stability is a rather imprecise term that refers to the propensity of the stream bank to change in form or location over time. In alluvial channels the stream and river banks tend towards a dynamic equilibrium with the discharge and sediment load. The bank material, vegetation type, and vegetation density also affect the stability and form of the streambanks (Platts, 1984). Change in any one of these factors is likely to be reflected in the size and shape of the stream channel, including the banks (Chapter 5).

Even in undisturbed streams some bank instability usually occurs. In valleys with a defined floodplain there is often lateral migration through bank erosion and point bar accretion (e.g., Leopold et al., 1964; Ritter, 1978). In V-shaped valleys there is less opportunity for lateral migra-

tion, and bank instability may stem from the input and eventual removal of obstructions emanating from fallen trees, landslides, or debris flows.

A higher incidence of bank instability can be initiated by natural events that disrupt the quasi-equilibrium of the stream, or by human disturbance. Extreme floods, wildfires, and landslides are three examples of short-term disturbances likely to affect channel form and bank stability. Climatic and tectonic change are two long-term processes that affect discharge, sediment load, and channel stability, but the time scale of these changes is well beyond the range of current water quality monitoring efforts. The ways in which human activities alter the discharge, sediment load, and streamside vegetation cover are discussed in Chapters 3, 4 and 6, respectively.

Relation to Designated Uses

Bank stability can be an important indicator of watershed condition and can directly affect several designated uses. Unstable banks contribute sediment to the stream channel by slumps and surface erosion. Because all the material from an eroding streambank is delivered directly into the stream channel, the adverse impact of bank instability can be much greater than the adverse effects of a comparable area of eroding hillslope.

Although in some cases the erosion of one bank will be matched by deposition on the opposite bank, streambank erosion caused by management activities generally will increase stream width. The corresponding increase in stream surface area allows more direct solar radiation to reach the stream surface, and this will raise maximum summer water temperatures (Sections 2.1, 5.2). In most cases an eroding streambank will provide little or no cover for fish.

Actively eroding streambanks also support little or no riparian vegetation, and the loss of this vegetation adversely affects a wide range of wildlife species (Raedeke, 1988), reduces available forage for domestic livestock, and reduces the long-term input of organic matter into the aquatic ecosystem. Both the increase in summer water temperatures and the loss of fish cover along an eroding streambank will be exacerbated by the reduction in riparian cover.

Response to Management Activities

The management activity that probably has the greatest impact on streambank stability is grazing (e.g., Platts, 1981). A reduction in the timing and intensity of grazing in the riparian zone often results in a decrease in channel cross-section, an increase in channel depth, and an increase in vegetation along the channel banks. All these changes suggest an increase in streambank stability, a reduction in sediment inputs into the stream channel, and an increase in the density of the riparian vegetation.

Increasingly stringent regulations have greatly reduced the direct adverse effects of forest management activities on those streams that have fish, are used for domestic water supply, or otherwise are granted a high level of protection. Small headwater streams and ephemeral channels generally do not have the same level of protection, and this can result in forest harvest and other management activities having a direct, adverse impact on bank stability. A large number of management activities can indirectly affect bank stability, as any change in the size of the larger (channel-forming) flows or in the size and flux of sediment is likely to alter channel morphology and hence bank stability (Sections 5.1-5.5).

Measurement Concepts

Standard procedures to evaluate bank stability have not been developed. Many stream monitoring programs focus on bank instability rather than bank stability, as eroding streambanks are often easier to identify and measure. Different monitoring programs have developed a variety of procedures to evaluate bank stability, and these range from qualitative, visual estimates to detailed measurements of each bank failure.

Perhaps the most widely used procedure related to bank stability is the method developed by Pfankuch (1978) to evaluate stream channel condition. This uses 4-6 parameters to evaluate the condition of the upper stream banks, the lower stream banks, and the channel bottom. These parameters are empirically weighted, and many of them are directly related to bank stability (Table 10). Summing the scores for all 15 parameters yields an overall rating for the stream channel (Pfankuch, 1978). Its use in the Pacific Northwest is sometimes criticized because it regards large woody debris as a destabilizing factor. Such comments do not demonstrate that the general method is faulty, but suggest that alterations in the parameters and scoring are needed as the technique is transferred to other areas and we gain an improved understanding of fluvial geomorphology.

A simpler procedure focusing solely on streambank stability is described in Platts et al. (1983, 1987). This technique assigns the bank along a specified cross-section to one of four stability classes according to the percentage of the bank covered by vegetation and rocks, and the size class of the rock material. The estimated percentage of the bank protected against fluvial erosion by rocks and vegetation provides a numerical rating of streambank stability.

A similar procedure can be used to determine streambank soil alteration (Platts et al., 1983; 1987). In this case the observer must visualize the appearance of the streambank under optimal conditions. The site is then assigned to one of five soil alteration classes according to the percentage of the streambank that has been broken down, eroded, or cut back from the stream. Again the actual percentage of the streambank that has been altered is estimated to yield a quantitative rating of streambank soil alteration. Platts et al. (1983, 1987) found that this technique had wider confidence intervals than the streambank stability rating, but the accuracy

Table 10. Parameters and range of values used for evaluating stream channel condition (Pfankuch, 1978).

Channel location Parameter	Range of values
Upper bank	
Sideslope gradient	0-8
Mass wasting potential	0-12
Debris jam potential	0-8
Vegetative cover	0-12
Lower bank	
Channel capacity	0-4
Bank rock content	0-8
Obstructions and flow deflectors	0-8
Bank cutting	0-16
Sediment deposition	0-16
Channel bottom	
Angularity of bed particles	0-4
Brightness of bed particles	0-4
Consolidation of bed particles	0-8
Stability and size of bed particles	0-16
Amount of scour and deposition	0-24
Aquatic vegetation	0-4

of both procedures could be rated as no better than fair. Errors were reduced when the rating was based on specific cross-sections rather than along a designated stream reach.

Standards

No standards for bank stability have been established or proposed.

Current Uses

Pfankuch's (1978) channel condition and stability procedure has been widely used by the U.S. Forest Service. Other monitoring programs have also taken elements from this rating system and incorporated them into their own stream evaluation forms (e.g., Ralph, 1989; G. Luchetti, pers. comm., King County, WA). Although the selection and weighting of the parameters have never been rigorously tested, the wide use of this procedure suggests a certain level of acceptance. One advantage is its accessibility to people with relatively little technical training, and it seems to provide relatively consistent results (Pfankuch, 1978). The arbitrary selection and weighting of parameters means that it should be modified according to local needs and experience, but this is rarely done.

Assessment

Streambank stability is an easily assessed parameter that can be used to indicate whether a particular stream has been disrupted from a quasi-equilibrium state. This disruption could be due to natural causes, or alterations in discharge, sediment load or vegetative cover caused by management actions (e.g., urbanization, grazing, forest harvest). Some of the major limitations to the use of bank stability include (1) lack of accuracy and precision (Platts et al., 1987), (2) inability to identify specific causes of bank instability (Platts et al., 1987), (3) varying sensitivity among stream reaches, and (4) difficulty of separating natural causes and management impacts.

The lack of accuracy and precision is partly a function of the techniques being used. The visual estimation techniques described by Platts et al. (1983, 1987) are likely to have greater uncertainty than the multi-parameter approach of Pfankuch (1978). One cannot conclude that a change in bank stability has occurred until the observed change significantly exceeds the error in the rating system, but this error is rarely recognized.

The cause of bank instability may be difficult to determine, particularly when there is more than one factor. Grazing has the most direct and obvious impact on bank stability (Platts, 1981), and this may mask other management impacts. Discharge and sediment yield tend to be controlled by upslope processes, and so the linkage to bank stability may not be immediately obvious.

Bank stability may be most useful as a quick indicator of a shift in the equilibrium of the stream system. An observed increase in bank instability should then trigger more intensive investigations. By combining an inventory of management activities with specific measurements of other parameters such as the bed material particle size, it is usually possible to determine the primary cause(s) of the observed disequilibrium. Often, however, bank instability may not be the most sensitive indicator of disturbance. Changes in the suspended sediment load, for example, may not immediately trigger bank instability, but could still have a detrimental effect on spawning success. Similarly, grazing impacts are likely to be expressed through the riparian vegetation before they lead to bank instability. Nevertheless, the ease of evaluating bank stability suggests that it can play an important role, particularly when budgets for assessment and monitoring are severely limited.

6. RIPARIAN MONITORING

INTRODUCTION

Characteristics of the riparian zone are rarely considered as water quality parameters, yet the riparian zone directly affects many of the designated uses of water. As noted in Sections 2.1 and 5.8, the type and amount of riparian vegetation is an important controlling factor for stream temperatures and bank erosion, and both temperature and bank erosion can be directly related to the quality of fish habitat. The riparian zone also plays a key role in defining channel morphology and creating fish rearing habitat through the input of large woody debris. Finally, the riparian zone is believed to be important in controlling the amount of sediment and nutrients reaching the stream channel from upslope sources.

Over the past 25 years, several major studies have documented the effects of forest harvest in the riparian zone on streams and water quality. The results of these studies have led to more stringent regulation of forest management activities adjacent to certain classes of streams (e.g., perennial streams a designated use of with coldwater fisheries or domestic water supply). The documented effects of management activities on the stability and vegetation of riparian zones, and the established linkages between the riparian zone and various designated uses, provide the rationale for including two riparian parameters in the *Guidelines*.

The first parameter is the width of the riparian canopy opening. Changes in the width of the riparian canopy opening generally result from changes in the balance between sediment and discharge. Hence the width of the riparian canopy opening may be a useful parameter for quickly determining historical trends in stream condition over large areas using aerial photographs.

The second parameter—riparian vegetation—is much more broadly defined. A variety of measurements can be made regarding the type and condition of the riparian vegetation, and these measurements may differ widely in their purpose, the amount of effort required, their sensitivity to different management activities, and their relation to the designated uses. The point is that the riparian vegetation and the width of the riparian canopy opening are important components of stream condition, and they can be useful parameters for monitoring the effects of management activities on streams.

6.1 RIPARIAN CANOPY OPENING

Definition

The riparian canopy opening refers to the gap between the canopy of the riparian vegetation on opposite banks of a stream or river. Often small streams are completely shaded by woody vegetation and hence have no riparian canopy opening in their undisturbed state. In steep, narrow, V-shaped valleys, considerable shading can result from the dominant upslope species rather than the riparian vegetation. In lower-gradient and higher-order streams, the stream channel by definition is wider and there commonly is a gap or opening between the parallel strands of the riparian vegetation. Streams with an alluvial valley floor tend to have more extensive and complex stands of riparian vegetation that develop in response to periodic flooding and high water tables.

These riparian and upslope forests that shade undisturbed stream channels can be altered by both natural disturbances (e.g., landslides, debris flows, and stream channel erosion) and forest management activities. Often a highly interactive response exists between changes in channel morphology and changes in the riparian forest (Wissmar and Swanson, 1990). For example, channel or bank erosion often changes the size and location of the stream channels, which results in a corresponding loss of the streamside

vegetation and an increase in the width of the riparian canopy opening.

Monitoring of the riparian canopy opening offers a relatively rapid means of assessing the influences of a variety of management activities on both the streamside vegetation and the stream channel. Identification of the source areas and quantitative mapping of the changes in the riparian canopy opening over time can help determine the primary cause(s) of adverse change (Grant, 1988).

Relation to Designated Uses

An increase in the width of the riparian canopy opening will allow more direct radiation to reach the stream and raise peak summer water temperatures. Less shading also will result in greater temperature fluctuations on both a seasonal and a daily basis (Section 2.1). A reduction in canopy cover may increase the amount of reradiated long-wave radiation, thereby allowing more heat loss at night. Heat loss can be crucial to the icing up and formation of anchor ice in colder environments (Beschta et al., 1987).

In light-limited forest streams, an increase in the width of the riparian canopy opening can increase primary production (Gregory et al., 1987). This may induce a corresponding increase in invertebrate and fish production. However, increased primary productivity may be offset by decreased inputs of detrital food subsidies, leaves, and other organic material from the riparian zone. The net balance between the increased primary production and the decreased detrital inputs will depend on the size of the stream and the presence or absence of other limiting factors, such as plant-available nutrients.

Changes in the size and structure of the riparian canopy will adversely affect a wide range of animal species dependent on riparian habitats (Deusen and Adams, 1989). A reduction in the width of the riparian zone may reduce the purported ability of the riparian zone to trap excess nutrients and sediments coming from upslope (Green and Kaufmann, 1989; Section 6.2). An increase in the riparian canopy opening is likely to reduce the long-term delivery of large woody debris (LWD) into the stream channel (Grant, 1988). In many forested streams LWD is an extremely important element in channel morphology, sediment transport, and quality of aquatic habitat (Bisson et al., 1987; Section 5.7).

Response to Management Activities

Changes in the size of the riparian canopy opening can result from a variety of interacting fluvial and geomorphic processes. Probably the most common cause is an increase in coarse sediment. This can increase channel width through bank erosion (Section 5.2), with a corresponding loss of the riparian vegetation. Recolonization of the enlarged streambed by riparian species will proceed slowly at best until the source of the excess sediment is removed, or the excess sediment in the channel is stored or transported out of the stream reach.

Grant (1988) noted that an increase in channel width and the riparian canopy opening also can result from an increase in the size of peak flows. As noted in Section 3.1, peak flow increases from forest activities usually are small or are limited to the smaller, more frequent storms. The major exception is in areas subject to rain-on-snow events; in these environments forest harvest can substantially increase the size of the larger peak flows (Section 3.1). In general, peak flows probably are less likely to enlarge the size of the riparian canopy opening and initiate channel morphological changes than increases in the amount of coarse sediment (i.e., bedload). Other possible causes of fluvial disturbance that can increase the riparian canopy opening include debris flows, extreme discharge events, entrainment and transport of large woody debris in flood plain areas, and increased lateral migration of stream channels.

Measurement Concepts

A detailed procedure for measuring and analyzing changes in the riparian canopy opening has been published as the RAPID (Rapid Aerial Photographic Inventory of Disturbance) technique (Grant, 1988). This requires a historical sequence of aerial photographs on a scale of at least 1:24,000. The basic approach is to (1) identify "initiation sites" where the increase in riparian canopy opening begins; (2) determine the spatial links between the initiation sites and downstream increases in the width of the riparian canopy opening; (3) determine the continuity of open reaches along the stream; and (4) measure the width of the riparian canopy opening and note the condition of the surrounding forest at 100- to 300-m intervals on each set of photos. These data are mapped onto drainage network maps at the same scale as the aerial photographs. Suggested procedures to analyze and summarize the quantitative data are presented by Grant (1988). Adaptations to the RAPID technique may be needed according to the specific vegetation, topography, geology, and climate in the basin under study.

Initiation sites are identified and mapped in order to elucidate the cause(s) of an increase in the riparian canopy opening. For example, a sudden, continuous increase in the width of the riparian canopy opening might be traced to a landslide or debris flow, whereas a more gradual increase in the width suggests a more dispersed source of sediment or an increase in the size of peak flows.

In many cases visual observations of the aerial photos will provide an indication of current condition, and riparian canopy opening measurements on successive aerial photos can demonstrate if adverse changes have occurred. The advantage of the full RAPID-type approach is that historical and current riparian and channel conditions can be quantified. This facilitates an understanding of the possible cause(s) of an observed change, an assessment of the significance of change, and the prediction of future trends.

As suggested above, the sensitivity of the riparian canopy opening to forest management activities will vary with stream type, location, and geological setting. For example, bedrock streams in steep, V-shaped valleys usually show little alteration in stream channel width in response to increased sediment load. Streams in wide valleys with unconsolidated alluvial sediments are likely to be much more sensitive to changes in flow and sediment flux (e.g., Lisle, 1982).

Standards

No standards have been established or proposed to regulate changes in the riparian canopy opening. However, most states have established Best Management Practices to restrict the removal of trees along fish-bearing streams or streams used for domestic water supply (Section 6.2). Leaving trees in or immediately adjacent to the stream channel helps maintain channel and bank stability, and therefore reduces the potential for an increase in the width of the riparian canopy opening.

Current Uses

Several studies have evaluated the concept of using the riparian canopy opening for monitoring. Grant (1986) showed a relationship between riparian canopy opening and area harvested for eight basins in or near the Middle Fork of the Willamette River, Oregon. A sequence of photographs for the Breitenbush River (Oregon) was used to document changes in the riparian canopy opening, and these changes were related to salvage logging and the December 1964 flood (Grant, 1988). In Western Washington the RAPID technique was used to identify the major disturbance events and changes in the riparian canopy, but it was not possible to directly relate disturbance events to downstream changes (Jeanette Smith, University of Washington, Seattle, pers. comm.).

A study of the Elk River Basin in southwest Oregon used the RAPID technique to document changes in the riparian canopy opening and relate these changes to timber harvest activities and large storm events (Ryan and Grant, in press). Upstream and downstream changes were not well synchronized, and this made it difficult to infer causal relationships. Currently, developmental projects designed to assess change in the riparian canopy have been initiated by the U.S. Forest Service and Washington State Department of Natural Resources.

Assessment

Determination of riparian canopy opening from aerial photographs appears to have considerable promise for quickly assessing stream condition and adverse management effects over a large area. These data can help assess the impact of past events and guide future activities. Monitoring of the riparian canopy opening is relatively unique because it uses an historic sequence of aerial photos as the primary data source. Such photos are available for most of the productive timberlands in the western U.S., and this permits change to be assessed over a period of several decades.

In contrast, most current water quality monitoring programs have only a few years of data. Any change observed during such a monitoring period cannot be placed in historical context, and this severely limits our ability to evaluate the significance of the observed change. Long-term data on the riparian canopy opening may provide some of the needed historical context.

On the other hand, the RAPID-type approach is not as sensitive to change as ground-based measurements. Unless unusually detailed aerial photos are available, an increase in the riparian canopy opening cannot be detected until a substantial increase in stream width has occurred. By this time much of the original banks and vegetation will have been lost, and some designated uses will have been impaired. In some cases it may be difficult to relate changes in the riparian canopy opening and the width of the stream channel to the potential causal factors such as landslides, forest harvest, extreme floods, or debris flows.

In summary, RAPID-type techniques have considerable potential for assessing change on a relatively broad temporal and spatial scale. This can help direct ground-based monitoring projects to the most critical locations, and provide a very useful context for the shorter-term data collected from such projects. However, as stream inventories or monitoring projects are initiated over larger areas and the data record is extended in time (Minshall et al., 1989; Gresswell et al., 1989), the need for a RAPID-type approach may gradually decline.

6.2 RIPARIAN VEGETATION

Definition

Riparian vegetation has been defined as "Vegetation growing on or near the banks of a stream or other body of water on soils that exhibit some wetness characteristics during some portion of the growing season" (AFS, undated). Other authors have specified that the soil should be saturated within the rooting depth of the plants for at least some portion of the growing season (Platts et al., 1983; Minshall et al., 1989).

These definitions suggest that riparian areas are a particular type of wetland. Wetlands have been defined by EPA as "Those areas that are inundated or saturated by surface or groundwater at a frequency and duration sufficient to support a prevalence of vegetation typically adapted for a life in

saturated soil conditions." A unified set of criteria for delineating wetlands has recently been adopted by several federal agencies with wetland responsibilities (U.S. Army Corps of Engineers, 1989). In most ecoregions one can find a wide variety of vegetation types on the streambanks and flood plains, including coniferous and deciduous trees, grasses, shrubs, forbs, ferns, and mosses.

Relation to Designated Uses

Riparian vegetation, and the exploitation of this vegetation, affects most of the designated uses of water through a variety of different processes. Many of these interactions have been discussed in other sections, and an extensive literature is available on the interactions between riparian zones and aquatic ecosystems (e.g., Raedeke, 1988; Gresswell et al., 1989). A recent bibliography on riparian research and management listed over 3,500 references (Van Deventer, 1990).

Some of the most important biological and physical effects of riparian vegetation on the designated uses of water are as follows: (1) providing organic material that can be used as food sources for aquatic organisms (Sections 7.3-7.4); (2) supplying large woody debris that alters sediment storage, influences channel morphology, and enhances fish production (Section 5.7); (3) shading the stream and reducing temperature fluctuations (Section 2.1); (4) reducing bank erosion (Section 5.8); and (5) providing habitat and cover for both aquatic and terrestrial organisms. Social benefits include streamside esthetics.

The relative importance of these different functions is heavily influenced by vegetation type. Deciduous trees provide large amounts of leaves and other organic material, which are generally higher in nitrogen than coniferous debris, and thus more readily broken down by invertebrates (Bilby, 1988). More rapid breakdown leads to more rapid utilization and higher productivity.

On the other hand, coniferous trees are the most important source of large woody debris in most parts of the Pacific Northwest and Alaska (Section 5.7). Coniferous branches, boles and root wads tend to be larger than their deciduous equivalents, and this increases both their stability within the stream channel and the diversity of aquatic habitats, particularly at high flows (Sedell et al., 1984; Bisson et al., 1987). Coniferous wood does not decay as rapidly as alder and most other deciduous species, and this also contributes to channel and habitat stability (Sedell et al., 1988).

Both coniferous and deciduous species are effective in shading the stream and thereby reducing peak summer temperatures. Streams with little or no vegetative canopy may have lower winter minima and be more susceptible to the formation of anchor ice (Platts, 1984).

All types of vegetation can be effective in reducing bank erosion, although they differ in the type of protection (e.g., Hackley, 1989; Platts and Nelson, 1989). Large trees and root wads can divert or deflect the flow in small or moderate-sized streams, and their roots can provide substantial protection during high flows. Grassy banks may provide a more complete cover, but they may not be as resistant to undercutting or abrasion.

Few studies have been done on the filtering and buffering capacities of riparian vegetation in forested zones (Green and Kauffman, 1989). In most undisturbed forest ecosystems the nutrient and sediment yields are so low that the filtering capacity of the riparian zone is not a key concern. In agricultural areas, however, nutrient exports are important and the riparian zone has been shown to be a sink for sediment as well as nitrogen, phosphorous, calcium, magnesium, and potassium sulfate (Lowrance et al., 1984; Lowrance et al., 1986; Green and Kauffman, 1989). The influence of different vegetation types on sediment and nutrient yields, and in some situations water yield, is complicated by differences in other factors such as the prevalence of overland flow, height of the water table, rooting depth, root densities, chemical properties of the soil, nitrogen-fixing ability of the plants, and seasonal growth patterns.

Various types of riparian vegetation provide different types of habitat (Raedeke, 1988). Species such as otters, beavers, deer, and bald eagles all have different habitat needs and are more or less dependent on riparian vegetation. Hence management of the riparian zone will depend in part on the selected wildlife and fisheries objectives. The uncertainty and subjective nature of habitat evaluations are illustrated by the observation that streams bordered by brush had a higher standing crop of fish than streams bordered by trees, yet the U.S. Forest Service usually assigns a higher habitat value to a tree cover (Platts et al., 1983).

The importance of the riparian vegetation to the adjacent aquatic ecosystem diminishes in the downstream direction because of the increase in discharge and stream size (Bilby, 1988). In small streams the riparian vegetation may be the dominant source of organic matter, while in larger streams instream primary production tends to dominate (Hynes, 1970). Removal or alteration of the riparian vegetation in a single reach can significantly alter temperature and water quality in low discharge, narrow streams, but the impact of a comparable change is likely to be undetectable in large streams or rivers (Bilby, 1988).

Response to Management Activities

The abundance of moisture makes the riparian zone exceptionally diverse and productive (Kauffman, 1988). This higher productivity often results in a more intensive exploitation of riparian resources. In many areas the largest trees are in the floodplains and alluvial valleys, and the riparian zones have been more heavily logged because the trees were readily accessible and could be floated downstream. Grazing pressure usually is higher in the riparian zone because there typically is more shade, surface water for

drinking, and more succulent vegetation (Platts, 1981). Riparian areas also tend to be the focus of recreational activities such as camping and fishing (Kauffman, 1988).

Several researchers have argued that livestock represents the single greatest threat to trout and wildlife habitat in the western U.S. (e.g., Behnke and Zarn, 1976; Platts, 1981). The inherent conflicts between livestock and fish make management of the riparian zone a more intractable problem in range lands than in forest lands.

The functions of the riparian zone described in the previous section provide the basis for predicting the effects of different management activities. Any reduction in the riparian canopy cover, for example, can affect stream temperatures, organic matter inputs, bank stability, and so on. A reduction in the riparian cover can result directly from management (e.g., harvesting trees in the riparian zone, grazing), or indirectly as a result of changes in the size and amount of sediment and discharge. As noted in Section 6.1, there are strong interactions between changes in the riparian vegetation and changes in stream channel morphology. The effects of management activities are reviewed in the sections on flow (Chapter 3), sediment (Chapter 4), channel characteristics (Chapter 5), and riparian canopy opening (Section 6.1).

Measurement Concepts

Although riparian vegetation affects many aquatic habitat and water quality parameters, generally it is more effective to monitor these other parameters directly rather than monitoring the riparian vegetation. Estimates of cover or rearing habitat for juvenile salmonids, for example, focuses on the type and abundance of cover in the stream, and not the potential cover, such as dead branches and snags, which may fall into the stream. Similarly, water quality parameters such as nitrate, conductivity, and turbidity are measured directly, and the influence of the riparian vegetation is difficult to assess. A notable exception is the increase in water temperature caused by removal of the riparian canopy. In short reaches with negligible groundwater flow, the increase in summer maximum temperatures is a direct function of the additional exposure of the stream surface to incoming solar radiation, and this effect can be predicted (Beschta et al., 1987; Section 2.1).

It follows that, with the exception of temperature, any precise measurement or characterization of the riparian vegetation provides an accuracy which cannot be translated into a more precise assessment of water quality or the impairment of designated uses. Thus relatively simple techniques that are repeatable over long time periods usually provide the best approach to monitor the condition of the riparian vegetation, and to evaluate the likely effects of the riparian vegetation on water quality (Platts et al., 1989).

An extensive literature on vegetation sampling is available, and the techniques for forests (e.g., Husch et al., 1982), shrublands, and grasslands (e.g., Cook and Stubbendieck, 1986; Tueller, 1988) can be applied as appropriate to the riparian zone. More often than not, however, stream inventory and water quality monitoring programs have developed *ad hoc* techniques for monitoring the riparian vegetation according to their particular objectives and conditions. The choice of qualitative or quantitative methods is determined by the parameter being measured, the anticipated use of the data, and the cost of collecting that data.

Some of the most commonly measured parameters include vegetation type, vegetation cover, and vegetation density. Vegetation type is usually a qualitative categorization which can be as simple as tree, shrub, grass or bare (e.g., Platts et al., 1983). More commonly the vegetation type is based on the dominant overstory species or specified plant communities (e.g., Platts and Nelson, 1989).

Vegetation cover usually refers to the downward projection of the canopy onto the ground surface (Husch et al., 1982). Visual estimation techniques can be used to provide a quick, qualitative measure (Platts et al., 1987). Quantitative measurements usually rely on point- or line-intercept methods.

Forest cover density can be assessed by measuring light intensity or by using a spherical densiometer (Lemmon, 1957). The latter uses a point sampling technique to determine the amount of clear sky in the hemisphere centered over the observer.

Data on stream shading can be obtained by several different methods. Sampling procedures for the spherical densiometer in large and small streams are discussed in Platts et al. (1987). Stream surface shading can be determined by measuring the height, density, crown width, and offset of the riparian vegetation. The Solar Pathfinder™ is a much simpler technique which directly maps the extent of shading on the specified day (Platts et al., 1987). Each of these techniques produces data useful for assessing changes in the riparian canopy over time, or for predicting the effect of riparian canopy removal on stream temperatures.

Vegetation density refers to the number of plants per unit area. For practical reasons density is most useful in forestry. It can be measured on either fixed- or variable-sized plots, and foresters often combine size and density data to obtain estimates of basal area or volume (Husch et al., 1982). Density, species, and size class data can be combined with growth and mortality data to estimate the future recruitment of large woody debris (e.g., Bilby and Wasserman, 1989).

Changes in the riparian vegetation due to grazing, logging, or other management activities can be assessed by each of these techniques. Cover, density, and biomass are more likely to reflect short-term management impacts than vegetation type. More frequent monitoring will be required in grazed areas owing to the rapid seasonal changes in forage availability and consumption. Platts et al. (1987) suggest a simple procedure to rate vegetation use in herbaceous areas.

Grazing strategies in riparian areas are discussed by Platts (1989), while recent books (e.g., Cook and Stubbendieck, 1986; Husch et al., 1982; Tueller, 1988) should be consulted for more details on vegetation monitoring techniques.

Standards

In the 1970s the forest practice rules for riparian areas in Oregon and Washington were designed to maintain adequate shade and minimize the introduction of sediment and forest chemicals (Bilby and Wasserman, 1989). Currently the rules for riparian areas are in a state of flux as a result of increased concern over the future recruitment of large woody debris into stream channels (Section 5.7). Present forest practice rules for Idaho require that 75% of the existing shade be left along Class I (fish-bearing) streams. In Washington the forest practice rules were substantially modified in 1988 under the Timber-Fish-Wildlife agreement, and the number of leave trees that currently are required along streams in eastern and western Washington is presented in Table 11. The complexity of the leave tree requirements in Table 11 illustrates the difficulty of trying to account for the diversity of natural systems in environmental regulations.

None of the state forest practice rules include any standards relating to grazing in the riparian zone, as that is outside their legal mandate. Land management agencies with substantial grazing lands have established utilization standards and Best Management Practices intended to protect the designated uses of water. The adequacy and evaluation of these is outside the scope of this document, but a recent publication by Minshall et al. (1989) and the proceedings of two recent conferences provide a good overview of riparian area functions and management (Raedeke, 1988; Gresswell et al., 1989).

Current Uses

The primary objectives of monitoring the riparian vegetation in forested areas are to maintain adequate shade,

Table 11. Type and number of trees required to be left along streams in western and eastern Washington after timber harvest (Bilby and Wasserman, 1989.)

Water type and average width[b]	RMZ maximum width	Ratio of conifer to deciduous	Minimum size of leave trees	Number of trees/ 305 m each side	
				Gravel/ cobble[a]	Boulder/ bedrock
1 and 2 waters 23 m and over	30 m	Representative of stand	Representative of stand	50 trees	25 trees
1 and 2 waters under 23 m	23 m	Representative of stand	Representative of stand	100 trees	50 trees
3 water 2 m and over	15 m	2 to 1	30 cm or next largest available[c]	75 trees	25 trees
3 water less than 2 m	8 m	1 to 1	15 cm or next largest available	25 trees	25 trees

[a]Gravel and cobble streambeds are composed predominately of material <25 cm in diameter.
[b]Washington water typing system is based on domestic water use, fish use, and size of streams. A detailed description of the criteria may be found in Washington Forest Practices Rules and Regulations.
[c]Or next largest available requires that the next largest trees to those specified in the rule be left standing when those available are smaller than the sizes specified. Ponds or lakes which are type 1, 2, or 3 waters shall have the same leave tree requirements as boulder/bedrock streams.
For wildlife habitat within the riparian management zone, leave an average of 12 undisturbed and uncut wildlife trees per hectare at the ratio of 1 deciduous tree to 1 conifer tree equal in size to the largest existing trees of those species within the zone. Where the 1 to 1 ratio is not possible, then substitute either species present. Forty percent or more of the leave trees shall be live and undamaged on completion of harvest. Wildlife trees shall be left in clumps, whenever possible.

Eastern Washington
(A) Leave all trees 30.5 cm or less in diameter breast height (dbh).
(B) Leave all snags within the riparian management zone that do not violate the state safety regulations.
(C) Leave 40 live conifer trees/hectare between 30.5 cm dbh and 50.8 cm dbh distributed by size as representative of the stand.
(D) Leave 7 conifer tree/hectare 50.8 cm dbh or larger.
(E) Leave the 5 largest deciduous trees/hectare 40.6 cm dbh or larger. Where these deciduous trees do not exist, and where 5 snags/hectare 50.8 cm in dbh or larger do not exist, substitute 12 conifer trees/hectare 50.8 cm in dbh or larger. If conifer trees of 50.8 cm dbh or larger do not exist within the riparian zone, then substitute the 5 largest conifer trees/hectare.
(F) Leave 7 deciduous trees between 30.5 cm and 40.6 cm dbh where they exist.
(G) On streams with a boulder/bedrock bed, the minimum leave tree requirement shall be 185 trees/hectare 10.2 cm dbh or larger.
(H) On streams with a gravel/cobble (less than 25.4 cm diameter) bed, the minimum leave tree requirement shall be 335 trees/hectare 10.2 cm dbh or larger.
(I) On lakes and ponds the minimum leave tree requirement shall be 185 trees/hectare 10.2 cm dbh or larger.

indirectly assess bank stability, and ensure an adequate supply of coarse woody debris. The first two are relatively straightforward, but the latter requires knowledge of tree growth rates, recruitment rates, the durability and stability of woody debris within a given stream, and specification of the desired amount of woody debris.

Tree growth rates for most areas are adequately known, and the durability and stability of large woody debris has been studied in a number of different streams (Section 5.7). Specification of the desired amount of woody debris is a political question. The last factor, recruitment rates, is the least known. Estimating recruitment rates is difficult because root wads, tree trunks, and large branches can enter into the stream channel by a variety of processes that vary both in magnitude and frequency (Bisson et al., 1987; Sedell et al., 1988). Small-scale, relatively frequent inputs include windthrow, natural mortality of trees along the stream channel, and bank erosion leading to the toppling of trees into the stream channel. Three more episodic mechanisms for delivering large quantities of woody debris to streams and rivers are debris flows, avalanches and landslides (Swanson and Lienkaemper, 1978).

The potential for each of these delivery mechanisms will vary with reach and catchment. In wide alluvial valleys the potential for episodic inputs is relatively small, and the future recruitment of large woody debris should focus on those trees which may fall directly into the stream channel. Streams with a rapidly migrating channel may have a wider recruitment zone.

The potential for large episodic inputs is greater in steep, unstable terrain, and areas subject to heavy falls of snow or rain. Large episodic events greatly expand the potential source area for large woody debris, but the frequency of these events is relatively low. Little or no data exists on the relative importance of the different processes for delivering large woody debris to the stream system in different catchments.

The methodology of Bilby and Wasserman (1989) provided a technical base for the Forest Practice Rules in Washington, but the procedures were not intended as a monitoring technique. Until better data is available on recruitment rates, monitoring to protect the designated uses of water will have to rely on measurements of large woody debris in streams and an assumed recruitment rate from trees immediately adjacent to the stream channel. An inherent limitation of these procedures is the long time frame needed to study the recruitment and stability of large woody debris in stream channels (Section 5.7). The management and silviculture of riparian zones is a primary focus of Oregon's COPE (Coastal Oregon Productivity Enhancement) program.

Assessment

Riparian vegetation is of critical importance to water quality because of its proximity to, and interactions with, aquatic ecosystems. In small streams the riparian vegetation usually is the largest source of organic material and hence a critical source of detritus for aquatic food webs. The riparian zone is also the primary source of large woody debris (Section 5.7). The amount of shade cast by riparian vegetation is an important factor in determining maximum stream temperatures and may also influence winter minima. Low overhanging riparian vegetation provides cover for salmonids and other fish (Platts et al., 1983). A reduction in the riparian vegetation through overgrazing, logging, or intensive recreational use can lead to bank erosion and instability. Bank erosion can have a disproportionate effect on water quality and the designated uses of water because the sediment is delivered directly into the stream channel (Section 5.8).

Monitoring of the riparian vegetation is another means of assessing management impacts in the riparian zone and evaluating whether certain designated uses are impaired. However, riparian vegetation cannot be used as a direct indicator of water quality except in the case of stream temperatures. For this reason most water quality monitoring programs use relatively simple, qualitative indicators to assess the type, density, and cover of the riparian vegetation. Detailed quantitative monitoring is most appropriate for

1. assessing stream shading and predicting the thermal effects of changes in the riparian canopy,
2. predicting the size and future recruitment of large organic debris;
3. measuring the amount of cover for fisheries, and
4. assessing bank stability and bank erosion as a function of vegetative cover.

Only the first of these cases can be classified as a traditional water quality parameter, even though the other three have clear linkages to water quality and some designated uses of water.

Management goals and the type of vegetation will largely determine the type of monitoring. In cool forested areas that are not heavily grazed, the emphasis should be on maintaining a healthy riparian canopy and ensuring adequate future inputs of large organic debris. In warmer areas stream surface shading is more likely to be the primary concern, and measurements of the riparian canopy can guide the intensity of management in the riparian zone. If grazing is the primary use, the emphasis should be on regular monitoring of bank stability (Section 5.8) and vegetative cover.

7. AQUATIC ORGANISMS

INTRODUCTION

Aquatic organisms can be very useful for monitoring because they effectively integrate a large number of habitat characteristics. In other words, if the habitat requirements of a particular organism are known, the presence of that organism can be used to define the conditions in that particular water body. Furthermore, those conditions can be assumed to have been met for the life span of the organism being monitored. Thus aquatic organisms have the great advantage of allowing inferences to be made regarding past conditions, which may allow sampling to be done less frequently than is usually necessary for the parameters considered in Chapter 2 (physical and chemical constituents) and Chapter 4 (sediment).

In this chapter aquatic organisms have been grouped into four parameters—bacteria, algae, invertebrates, and fish. Bacterial monitoring is the most straightforward and typically involves estimating the numbers of up to four types of bacteria. Algae, invertebrates, and fish are far more complex, as one can make any number of measurements relating to the numbers of organisms, species composition, and productivity. These different measurements are not considered separately for several reasons.

First, we did not wish to duplicate the considerable amount of information already available on the use of these organisms for monitoring. Second, there is no consensus about which measurements should be made, and in many cases the choice will depend on the purpose of the monitoring. Third, the use of aquatic organisms for monitoring is undergoing rapid change as different states attempt to establish biological criteria for water quality. Fourth, the tremendous variability in aquatic ecosystems has made it difficult to establish rigorous and sensitive monitoring procedures. Finally, the use of fish for monitoring purposes is often hindered by the problem of separating extraneous factors, such as fishing pressure, from the effects of management activities.

In general, aquatic organisms have considerable potential for monitoring changes in water quality. Aquatic invertebrates are particularly promising because of their diversity, sensitivity to habitat change, relative ease of identification, and they are subject to fewer extraneous controlling factors. The complexity and diversity of aquatic ecosystems means that the sections on algae, invertebrates, and fish should be considered as a general overview rather than an in-depth review.

7.1 BACTERIA

Definition

A wide variety of diseases are spread by aquatic microorganisms. These include bacterial diseases (e.g., Legionnaire's disease, cholera, typhoid, and gastrointestinal illness), viral diseases (e.g., polio, hepatitis, and gastrointestinal illness), and parasitic diseases (amoebic dysentery, flukes, and giardiasis). Many of these diseases are rarely found in the U.S., and the analytic procedures for detecting many of these organisms are time consuming and costly. For these reasons most drinking and recreational waters are routinely tested only for certain bacteria which have been correlated with human health risk. If the average concentration of these bacteria falls below the designated standard, it is assumed that the water is safe for that use and that there are no other pathogenic bacteria that represent a significant hazard to human health (APHA, 1989).

The four groups of bacteria most commonly used for water quality monitoring are total coliforms, fecal coliforms, fecal streptococci, and enterococci. The total coliforms (TC) group includes a wide range of aerobic and facultatively anaerobic bacteria. Their ability to ferment

lactose and produce gas helps define the group and also is the basis for one of the primary testing methods. Many coliform bacteria are non-pathogenic and are not associated with human waste.

Fecal coliform (FC) bacteria are mostly those coliform bacteria which are present in the gut and feces of warm-blooded animals. The primary species in this group are *Escherichia coli* and *Klebsiella* species. They are distinguished by their ability to produce gas from lactose at a temperature of 44.5±0.2°C. Generally they are less able to survive in natural waters than non-fecal coliform bacteria.

Fecal streptococci (FS) also are found in the intestines of man and animals, but in animals FS is usually more common than FC. This observation has led to efforts to use the FC/FS ratio to determine whether contamination is due to man, animals, or a mixture of the two; however, a number of restrictions on the use of the FC/FS ratio exit (EPA, 1978). One problem is that FS and FC have different die-off rates in natural waters, so the FC/FS ratio is useful only for the first 24 hr after contamination has occurred. The more limited ability of some FS species to survive in natural waters indicates that FS concentrations should not be the sole test of bacterial contamination.

The enterococcus group of bacteria is part of the larger FS group. These bacteria are of particular interest for monitoring recreational waters because they appear to be a better indicator of the risk of gastrointestinal illness than TC, FC, or FS (Vasconcelos and Anthony, 1985).

Relation to Designated Uses

The concentration of TC bacteria has long been used as the primary criterion for the sanitary condition of domestic water supplies. Experience has repeatedly demonstrated a positive correlation between the TC count and the incidence of gastrointestinal disease. However, many of the TC bacteria do not have a direct effect on human health and are found outside of animal intestines and feces. This means that TC are not a particularly accurate or consistent indicator of the actual health risk.

FC, FS, and enterococci are more specialized groups of bacteria than TC. Their more restricted habitat means that their concentration can be more directly linked to sanitary water quality and human health risks. FC have been considered a better indicator of water quality than total coliforms for over 2 decades. More recent evidence indicates that enterococci concentrations are most closely correlated with gastroenteritis among swimmers (DuFour, 1982). *Escherichia coli* was the next best indicator, while the broader group of FC were a relatively poor indicator of health risk. Both FC and FS, although less tolerant of the aquatic environment than most other types of coliform bacteria, can survive for several days in fresh water.

The public generally is aware of the significance of coliform bacteria in indicating water quality. Severe adverse public reaction can be expected if recreational or domestic waters do not meet bacteriological standards.

Response to Management Activities

High counts of TC, FC, or FS usually are associated with inadequate sewage treatment, poorly functioning septic tank drainfields, or high concentrations of animals. In forested areas, high levels of coliform bacteria usually will be associated with inadequate waste disposal by recreational users, the presence of livestock or other animals in the stream channel or riparian zone, and poorly maintained septic systems. Since each of these is a relatively dispersed source, and the soil is an excellent filtration medium, bacterial contamination can be greatly reduced simply by locating these activities away from the stream or lake boundary (Kunkle et al., 1987). However, septic systems may not function effectively in cold climates or in certain soil types.

Measurement Concepts

The two most common methods for measuring TC, FC, and FS are the multiple-tube fermentation technique and the membrane filter technique. The multiple-tube fermentation technique places varying amounts of the sampled water in tubes containing a growth media. These tubes are incubated for up to 48 hr to determine if gas bubbles form; gas formation is regarded as a sign that coliform bacteria are present in that sample ("presumptive test"). Tubes testing positive may be subjected to additional procedures to confirm the presence of coliform bacteria ("confirmed test" and "completed test"; APHA, 1989).

The two main problems with the multiple-tube fermentation test are as follows: (1) an individual tube indicates only whether coliform bacteria are present or absent in that particular sample, and (2) the false positive rate for a single tube is 13% (Federal Register, 1989). For this reason at least five replicate tubes at several different dilutions now are required to obtain a reliable estimate of bacterial concentration (APHA, 1989).

Replication is needed to provide a higher level of confidence in the results. If the true concentration of coliform bacteria is one organism per ml, for example, there is a 37% chance that a well-mixed, 1-ml sample will not have any coliform bacteria (APHA, 1976). If five tubes, each with a 1-ml sample, are tested, there is a less than 1% chance that all five tubes will yield a negative result. Replications also assist in making a quantitative estimate of the coliform concentrations, as a properly diluted sample will have a mixture of positive and negative results. For this reason replications generally are required when testing for coliform bacteria (Federal Register, 1989).

Different dilutions are needed to obtain a quantitative estimate of bacterial concentrations. Ideally the range of dilutions will span the range of results from nearly all

positive to nearly all negative. The replications and range of dilutions allow a statistically based estimate of the coliform concentration for that particular sample, and this is known as the Most Probable Number (MPN). The binary nature of the procedure (i.e., each tube is either positive or negative) results in a relatively large confidence interval around the MPN. For five replicates at three dilutions, the 95% confidence interval usually spans a factor of 10. The use of only three replicates doubles or triples the size of the 95% confidence interval, and this is why the minimum number of replicates has been raised to five tubes (APHA, 1989).

The second analytic procedure is the membrane filter technique. In this method different volumes of the sample water are passed through a special 0.45-micron filter. The filter and retained bacteria are placed on a selective growth medium and incubated for 24 hr. At the end of this period, the actively growing, closed coliform colonies are identified and counted. For best results the quantity of water filtered should yield about 50 colonies for TC, and between 20 and 60 colonies for FC. No more than 200 bacterial colonies should be present on one membrane filter (APHA, 1989).

There are several advantages to the membrane filter test. First, the filtering procedure allows for larger volumes of water and, hence, more accurate testing of less polluted waters. Second, the results are more precise and have a lower confidence interval because each sample yields a quantitative result. Third, the procedure yields results within 24 hr, although additional testing may be necessary for further verification. A major disadvantage is that the test can be hindered by high concentrations of either suspended solids or non-coliform bacteria (APHA, 1989; Federal Register, 1989).

In mid-1989 EPA approved a third analytic procedure for total coliforms in finished drinking water, the MMO-MUG test. Conceptually this is similar to the fermentation tube technique, except that the end result in the MMO-MUG test is a change in color rather than the production of gas. The MMO-MUG test may prove more convenient because the incubation period is only 24 hr, and it is not affected by large numbers of heterotrophic bacteria. MMO-MUG tubes with the growth medium are commercially available.

Standards

The drinking water criteria for TC is zero with some allowance for an occasional positive test.

For freshwater bathing, the geometric mean value of at least five samples equally spaced over a 30-day period should not exceed 126 *E. coli*/100 ml, or 33 enterococci/100 ml (EPA, 1986b). The maximum value for any single sample is determined by the intensity of recreational use and the site-specific standard deviation of the logarithmic values. Thus the allowable maximum for a single sample will be higher in areas which are infrequently used for bathing, and higher in areas which are subject to more variability in bacterial counts. This approach means the standards are based partly on the relative health risk rather than an absolute standard.

Current Uses

Bacteriological testing is regularly carried out to ensure the safety of domestic water supplies and to protect public health in recreational areas. Most states have adopted FC as the primary standard for bacterial contamination in recreational waters and test accordingly. As indicated above, the standards include both single sample maxima and a 30-day average. FC are preferred over TC because they are a more specialized group of bacteria and a more direct indicator of fecal contamination and public health hazard. Use of enterococci for monitoring recreational waters is becoming more common because this test is more sensitive and provides a better estimate of the human health risk.

TC, FC, and FS concentrations often vary widely over relatively short time periods. For this reason any monitoring or assessment program should analyze a series of samples before coming to a conclusion about the bacterial quality of the water. In most cases, the maximum concentration of coliform bacteria will occur in conjunction with high runoff events, which wash more coliforms into streams and lakes. Common sources are manure from animals and bypass water from small community sewage treatment plants. In still waters used for bathing, the maximum concentration of coliform bacteria may occur during warm-weather periods when there is intensive use.

Very specific tests can be performed to identify the different species of coliform bacteria, and this information can help identify the source of the contamination. Since these tests are relatively costly and not widely available, the source(s) of contamination usually is identified by establishing a more intensive sampling program keyed to land use. Use of the FC/FS ratio is cautioned because of the different mortality rates and sources of these two groups in natural waters.

Assessment

Bacterial contamination is the only water quality monitoring parameter discussed in these *Guidelines* that has little effect on aquatic organisms, but is very significant to human use. Bacterial contamination in forested areas can result from a variety of sources, including dispersed and developed recreation, wild and domestic animal populations, and human settlements.

The use of bacterial parameters to monitor water quality for drinking and bathing is based more on correlations than a direct causal link. Historically, total coliforms have been used as the primary bacterial indicator of human health risk; however, over the last 20 years, three more specialized groups of bacteria have been increasingly utilized for water

quality monitoring because they show a better correlation with human health risk. The three groups for which procedures (APHA, 1989) and standards (EPA, 1986b) have been adopted are fecal coliforms, fecal streptococci, and enterococci. Each of the four groups currently used in water quality monitoring has a particular significance, and the use of more than one group may be beneficial in some cases.

Total coliforms are useful for assessing contamination of finished drinking water because they are the largest and most diverse group, and any bacterial contamination of drinking water is considered unacceptable. Fecal coliforms are a better indicator of contamination in natural waters, and this is largely due to their more specialized nature. Fecal streptococci are similar to fecal coliforms, but more common in animals than fecal coliforms. In some cases the ratio of fecal coliforms to fecal streptococci can provide some insight on the source(s) of contamination, but this ratio must be applied with caution (EPA, 1978). Streptococci are the most recent addition to the family of bacterial parameters, and this appears to be the best indicator of contamination for recreational waters because it is both more sensitive and more directly correlated with human health risk (e.g., Vasconcelos and Anthony, 1985).

Bacterial counts tend to be highly variable over time, and the standards for drinking and bathing explicitly recognize this variability. The standards for bathing waters also recognize that the link between bacterial counts and human health is indirect. Thus waters used infrequently for bathing have less stringent standards than waters at designated bathing beaches.

This means that the type and frequency of monitoring for bacterial contamination should depend on the beneficial use. More intensive monitoring is appropriate in areas which provide domestic water supplies, or in areas which have heavy recreational use. In these cases any sign of contamination is likely to require an immediate management response and public notification. On the other hand, the high variability of bacterial counts means that any single test is of questionable value, and this is particularly true for total coliform.

The above considerations suggest that one should err on the side of caution when designing a bacteriological monitoring program and analyzing the resulting data. The relatively high cost of not detecting contamination and the relatively low cost of analyzing individual samples mean that monitoring should be more regular and intensive than for most of the other monitoring parameters discussed in these *Guidelines*. For the same reasons any statistical analysis might use a larger alpha value (i.e., a greater likelihood that the results are due to chance) in exchange for more power (i.e., a greater likelihood of finding contamination when it is present) (Part I, Section 3.4.2).

7.2 AQUATIC FLORA

Definition

The flora responsible for primary production in aquatic environments can be classified taxonomically, functionally, or morphologically. In classical plant taxonomy, the primary groups of aquatic plants are the algae, vascular macrophytes, and mosses. In most streams and lakes in forested areas, the bulk of the primary productivity is due to algae (Hynes, 1970).

Aquatic ecologists often use a functional classification with three primary categories: (1) free-floating or planktonic forms, (2) plants attached to the substrate, and (3) plants rooted into the substrate (Weitzel, 1979). The relative importance of these three categories is determined largely by the physical features of the habitat. Free-floating plants, for example, are significant only in still waters or large rivers where there is sufficient time for them to build up their populations. Rooted aquatic plants are rarely found in areas where the bed material is coarse or subject to frequent transport. Attached plants—mainly benthic algae—are most important in gravel-bedded headwater streams. Some streams may receive free-floating plants washed in from lakes or backwater areas (Hynes, 1970).

Morphologic classification systems for aquatic flora can be simpler than the taxonomic and functional approaches. The usual distinction is between microflora and macroflora, but these are arbitrary size classes, and in the initial growth stages macroflora species can be part of the microflora (Hynes, 1970).

Most studies of aquatic flora have concluded that the attached plant community is best suited to water quality monitoring (Weitzel, 1979). Two terms are commonly used to refer to the attached flora—Aufwuchs and periphyton. Although some authors consider these synonymous, Aufwuchs—a German term meaning attached growth—refers to all organisms growing on or attached to a substrate, and this includes heterotrophic organisms such as bacteria, bryozoa, and sponges, as well as small mobile organisms (e.g., protozoans and insect larvae) living within the mat (Power et al. 1988; Ruttner, 1953; Wotton 1988). Periphyton often has a slightly narrower definition—aquatic flora growing on submerged substrates—and this may or may not be limited to the microflora (Cattaneo 1987; Hutchinson, 1975; Odum, 1971; Weitzel, 1979). In forested streams in the Pacific Northwest, the attached algal communities are commonly referred to as benthic or epibenthic algae (Hudon and Legendre 1987). Diatoms usually are the most important and diverse algal group in benthic communities (Pryfogle and Lowe, 1979). Epiphytic algae refers to attached microalgae (e.g. diatoms) that grow on the surface of macrophytes (Cattaneo and Kalff, 1980).

Relation to Designated Uses

Benthic algae can be the dominant group of primary producers (photosynthetic organisms) in stream ecosystems (Hynes, 1970; Wetzel, 1983). Mats of attached algae form rich assemblages of plant, bacteria, and animal species, all of which are important components of the overall food web (Weitzel, 1979; Power et al., 1988). In small headwater streams, the contribution of organic matter by benthic algae may be outweighed by inputs of organic matter from riparian and forest vegetation. With increasing stream size, however, the importance of autotrophic production increases. Increased benthic algal production is linked to increased production of benthic invertebrates and fish (Gregory et al., 1987).

In lakes and downstream portions of slow-flowing rivers, all three functional plant groups—free-floating, attached, and rooted—can affect the designated uses of water and be ecologically important habitats (Power et al., 1988). High levels of free-floating plants, for example, will impair the clarity of the water and may have adverse esthetic effects. Aquatic macrophytes can adversely impact recreational uses such as swimming and boating, and also degrade the esthetic value.

Ecologically, an increase in primary production can increase the production of invertebrates and fish in streams. However, nocturnal respiration can cause oxygen depletion in waters with high primary production and low reaeration rates. Even relatively small reductions in dissolved oxygen can have adverse effects on both invertebrate and fish communities (Section 2.4). Development of anaerobic conditions will alter a wide range of chemical equilibria, and may mobilize certain chemical pollutants as well as generate noxious odors.

High primary production also can lower the concentration of nitrogen and phosphorus because of the rapid uptake of nitrate, ammonium, and phosphate by algae and other aquatic plants (Section 2.5). Aquatic plants can influence the color, taste, and odor of water (APHA, 1976).

Response to Management Activities

Numerous studies have related organic pollution to specific aquatic plants or plant community parameters. Relatively little definitive data are available on the effects of forest management activities on aquatic plants. Specific activities that might be expected to affect aquatic plants include herbicide applications, opening up of the riparian canopy, increased stream temperature, increased nutrient concentrations, and sedimentation.

Aerial herbicide applications may adversely affect primary productivity, but this is highly dependent upon the protective measures taken. The use of buffer strips, appropriate application technology, and good weather conditions can greatly reduce the amount of herbicide reaching the stream channel. Sullivan et al. (1981) found no toxic effects on stream and pond benthic algae following the application of a herbicide (Roundup) in coastal Oregon. In the coastal Carnation Creek watershed in British Columbia, Holtby and Baillie (1989) observed a decline in benthic algal standing crop and biomass accumulation in the first month after a glyphosphate application. In both studies the large temporal and spatial variability in algal growth and abundance made it difficult to determine the effect of the herbicides.

Partial or complete removal of the riparian canopy will increase direct solar radiation, and this may increase benthic algal growth. In headwater streams of the Cascades, primary production is proportional to sunlight at low light intensities. At 20% of full sunlight, the benthic algal communities are photosynthetically saturated, and additional sunlight may not enhance production (Gregory et al., 1987).

The temperature increases associated with forest harvest and sedimentation (Section 2.1) affect primary production and respiration. In general, an increase in temperature will increase the rate of respiration more rapidly than the rate of photosynthesis, so an increase in temperature decreases net primary production (Gregory et al., 1987). In most cases the effects of a change in temperature cannot be detected, as a laboratory study showed that primary production increased by only 30% following a 10°C increase in temperature (Gregory et al., 1987). High light intensities appear to favor filamentous green algae, and this may explain the observed increase in abundance following clearcutting (Stockner and Shortreed, 1988).

As discussed in Section 2.5, a variety of forest management activities can increase the availability of nitrogen and phosphorous, and this has been demonstrated to stimulate primary production (e.g., Gregory, 1980; Stockner and Shortreed, 1978; Triska et al., 1983). Increased stream productivity, due to increased nutrient output from watersheds following harvest, typically lasts only a few years (Gregory et al., 1987; Vitousek et al., 1979). The rapid uptake of nutrients by primary producers means that increases in production may be quite localized (e.g., Holtby and Baillie, 1989).

Increased sedimentation can reduce primary production by reducing the area of suitable substrate and by reducing the depth of light penetration. The most damaging sediment is sand-sized particles, as they are easily mobilized and do not provide an adequate surface for colonization (Hynes, 1970). Increased bedload may increase primary production by increasing stream width and temperature (Section 4.3). An increase in silt- and clay-sized particles will tend to decrease primary production by reducing the amount of light within the water column and coating the stream bed (Section 4.2).

This discussion indicates that forest management activities affect the productivity and composition of the aquatic flora in different ways by a variety of processes. The net effect will depend on the relative balance and interactions

among these effects. In most cases the net change in the aquatic flora can be linked to some of the designated uses of water. Usually, however, the spatial and temporal variability in the aquatic flora will preclude the definitive detection of management effects in streams, and hence the impact on designated uses can only be assumed. For example, benthic algae in many streams undergo dynamic cycles of growth, senescence, decay, and export. Although information is available about some factors that influence algal biomass at a site, little is understood about the effects of factors such as floods or algal grazing by aquatic organisms.

Measurement Concepts

Of all the aquatic plants, algae have long been the most widely used indicator of water quality and stream condition (Hynes, 1966; APHA, 1976; Weitzel, 1979). Some advantages of using algae include the following:

1. Their presence and growth integrate numerous physical factors;
2. their relatively short life cycle makes them useful indicators of short-term impacts;
3. they are sensitive to certain pollutants, such as herbicides and excessive inputs of nutrients, which may not affect other organisms;
4. sampling can be easy and inexpensive depending on the situation; and
5. relatively standard methods exist for evaluating the structural and functional characteristics of algal communities (EPA, 1989).

Disadvantages to the use of algae and other aquatic plants are as follows:

1. They are highly variable with location (Pryfogle and Lowe, 1979);
2. they are highly sensitive to small changes in current velocity, substrate type, and other physical factors (Weitzel et al., 1979);
3. considerable expertise and time are needed to identify both attached and free-floating microflora species; and
4. the use of qualitative information, such as the presence or absence of particular species, may be invalid or appropriate only on a very coarse scale (Weitzel, 1979; Weitzel et al., 1979).

Both species and community parameters have been used to characterize aquatic plants and monitor water quality. The simplest technique is to use selected species as indicators of water quality. This assumes that the habitat requirements of a particular species are known, that the habitat requirements are relatively constant, and that presence or absence is solely a function of water quality. Lists of species associated with organic pollution have been developed and used to distinguish up to nine different zones of pollution (APHA, 1976; Weitzel et al., 1979). The indicator species approach is limited in that it allows only a qualitative assessment of stream condition from specific pollutants, and it has been widely criticized (Patrick, 1973; Pielou, 1975; Platts et al., 1983).

Community parameters can be divided into structural characteristics, such as species richness, diversity, or biomass, and functional characteristics, such as productivity (Odum, 1971; Rodgers et al., 1979). For benthic algae, these parameters can be measured from natural or artificial substrates.

Artificial substrates generally are accepted as being comparable to natural substrates (Weitzel et al., 1979), and their use eliminates the variability due to substrate type. Several studies have shown that the variation between replicates for parameters such as chlorophyll-a and ash-free dry weight typically is 20-25% (Weitzel et al., 1979). Much larger differences were found between artificial substrates placed in apparently similar locations, and this was ascribed to small differences in current velocity and solar radiation (Weitzel et al., 1979).

The values and limitations of species richness and diversity data are discussed in conjunction with benthic invertebrates (Section 7.3), as they have been studied more intensively than the aquatic flora. Patrick (1973) asserts that diatoms are well suited to monitor pollution, but her methodology requires counts of 5,000-8,000 individuals per site. Other studies have used smaller counts, and the available data suggest that at least 500 organisms are needed to estimate the species distribution (Weitzel, 1979). Normal procedures for fixing diatoms leave only the frustules (shell), and this precludes the separation of live and dead diatoms. Inclusion of dead diatoms in estimates of community parameters can substantially bias the results (Owen et al., 1979). Other floristic groups, such as macrophytes or phytoplankton, usually are too location-specific or too rare to use for estimating community parameters.

Biomass refers to the organic matter content per unit area or volume, and this is sometimes incorporated in monitoring programs. A correlation between water quality and biomass is difficult to establish because so many other factors, such as light, nutrients, and grazing intensity, may be limiting (Weitzel et al., 1979). Another problem with the use of biomass data is that up to 80% of the dry weight of benthic algal communities is composed of sediment, diatom frustules, and other inorganic matter that accumulates in the algal mat. For this reason biomass estimates should always be based on the ash-free dry weight (Weitzel et al., 1979).

Chlorophyll-a is often used as a surrogate for biomass. Typically the amount of chlorophyll-a is 1-2% of the ash-free dry weight, but values can range between 0.15 and 4% (APHA, 1976; Weitzel et al., 1979). Factors affecting the concentration of chlorophyll-a include the age and physiological state of the organism, amount of dead biomass present, community composition, and abiotic factors such as light intensity and nutrient availability (Clark et al., 1979; Hudon and Legendre, 1987).

The Autotrophic Index is the ratio of ash-free dry weight to chlorophyll-a. A value less than 50-100 indicates that virtually all the periphytic organisms are algae that are actively photosynthesizing (autotrophs), and that there are few organisms utilizing organic matter and pollutants (heterotrophs). Values higher than 100-200 indicate that a substantial proportion of the biomass is composed of organisms that are not photosynthesizing (APHA, 1976; Weitzel, 1979). However, ratios of 200-400 for actively growing filamentous assemblages have been observed under laboratory conditions (S. Gregory, Oregon State Univ., pers. comm.). Hence the Autotrophic Index is potentially a useful ratio but may have limited applicability when the primary pollutants are not rich in organic matter.

The primary metabolic processes of aquatic plants are primary production (photosynthesis) and respiration. Neither of these is easily measured, particularly in stream systems where the flow of water is of critical importance (Weitzel, 1979; Wetzel, 1983). In most cases an index that approximates production can be obtained by measuring the accumulation of organic material (e.g., biomass) on artificial substrates over a period of 1-2 weeks. Other factors besides water quality may affect production, respiration, and the net rate of biomass accumulation; these include grazing, sloughing, scour, colonization, and deposition. Hence a high turnover rate (primary production divided by biomass) can result in a low rate of biomass accumulation but a high rate of primary production. In the absence of this type of detailed information, it is difficult to relate water quality to either algal production or biomass.

Standards

No specific standards have been established or proposed for aquatic plant communities, although an objective of the Clean Water Act is to restore and maintain the biological integrity of water bodies. More specific biological criteria are now being developed by the states (Part I, Section 1.4; EPA, 1988b; EPA, 1990).

Current Uses

The use of aquatic plants other than benthic algae for monitoring water quality may be more appropriate in lakes. In lakes, both free-floating plants and aquatic macrophytes may be directly linked to specific designated uses. Thus an observed increase in algal biomass or production can not only indicate a change in water quality but also can be related to a designated use, such as recreation. In streams, however, benthic algae production and biomass probably are the most useful of all the aquatic flora parameters to monitor changes in water quality. In both streams and lakes, the two main problems with monitoring aquatic plants are (1) detecting a statistically significant change in the face of large spatial and temporal variability, and (2) relating any observed change to specific management activities.

The first problem is a sampling problem. It can be addressed by specifying the trade-offs between sampling costs, the risk of an erroneous result, and the probability of obtaining the true answer. A small pilot study is often needed to adequately evaluate these trade-offs (Part I, Chapter 3). Long-term data are necessary to determine if an observed change is either part of a larger trend or within the range of previous changes.

The second problem may be more difficult. In most cases a variety of additional data (e.g., nutrient concentrations and incoming solar radiation) will be needed to determine the cause of observed change. An increase in biomass or chlorophyll-a, for example, could be caused by an increase in nutrient levels, warmer temperatures, or a reduction in grazing. Data on management activities within the watershed usually are necessary to determine the likely cause(s). In most cases the results will not be definitive, and some extrapolation or assumptions will have to be made.

Assessment

Benthic algae and attached algae on large macrophytic plants (epiphytic algae) can dominate primary production in many streams and rivers and provide the main source of organic matter. Attached algae provide both food and habitat for a wide range of invertebrates, and these invertebrates are an important source of food for salmonids and other fish (Power et al., 1988).

In lakes free-floating plants and macrophytes may be of primary importance. Species composition, biomass, and productivity of aquatic plants have been used to indicate up to seven different levels of lake eutrophication. Such detailed determinations usually are based on the identification of large numbers of diatoms, and this generally precludes their use in most monitoring projects. These procedures also may be of limited applicability in forested areas because there typically is very little eutrophication, and the applicability of the techniques to oligotrophic systems has not yet been established.

Attempts to relate forest management activities to the composition and growth of benthic algae have met with limited success. The variability associated with replicated artificial substrates (glass slides) within a sampler is 20-25%. Differences between samplers placed in "similar" environments are much greater, and this severely limits one's ability to detect statistically significant change over time or space. Although aquatic plants can be directly linked to several designated uses, it usually is better to measure the causative factors (e.g., changes in temperature, riparian canopy opening, or bed material particle size) rather than the resulting change in benthic algae or other aquatic plants.

Aquatic plants are more likely to affect the designated uses of water in lakes than in streams. In both stream and

lake ecosystems, some algal indicator such as chlorophyll-a is generally the most appropriate monitoring technique. The collection of presence/absence, species richness, and species diversity data all require a trained taxonomist and may require the identification of a large number of microorganisms. The cost and difficulty of carrying out such a program has led most people to use some indicator such as the concentration of chlorophyll-a. This is a useful approximation of algal abundance, but it is not sensitive to small changes. Artificial substrates are unlikely to provide greater sensitivity, and their use is advantageous only if other parameters, such as ash-free dry weight or species composition, are to be obtained from the samples. The area covered by aquatic macrophytes might be another useful indicator of river or lake conditions. No single technique is optimal under all situations, and additional data will be needed to identify the most likely cause(s) of an observed change in the aquatic plants.

Even though the value of aquatic plants for water quality monitoring may be limited, any data will increase our understanding of the aquatic system. A measurement of instream primary production, for example, may provide some indication of the likely response of the algal community to nonpoint source pollutants like nutrients and sediments, even though we may not be able to directly measure this response. Such information also could help indicate the relative balance between primary production and terrestrial organic inputs, and this information could help guide riparian zone management. In general, an increased understanding of the structure and functions of aquatic ecosystems should improve both management effectiveness and the protection of aquatic resources.

7.3 MACROINVERTEBRATES

Definition

Macroinvertebrates are animals without backbones that are large enough to be seen with the naked eye. The lower size limit is arbitrary. The U.S. Geological Survey has adopted a mesh size of 0.21 mm as the most suitable for sampling macroinvertebrates in flowing waters (Platts et al., 1983), while APHA (1989) defines macroinvertebrates as those invertebrates retained on a U.S. Standard No. 30 sieve (0.595 mm openings).

A wide variety of taxonomic groups are found in freshwater environments, and these include annelids, crustaceans, flatworms, mollusks, and insects. Benthic macroinvertebrates, which live on the stream bottom, are the group most amenable to systematic study. Most research has focused on aquatic insects, as these are the most common and diverse macroinvertebrates in forested areas. It follows that most freshwater monitoring programs have been directed towards benthic aquatic insects, and these organisms will be the primary focus of this section.

Relation to Designated Uses

Macroinvertebrates play several major roles in aquatic ecosystems. They graze on periphyton (attached algae) and feed on the terrestrial organic material that falls into the stream. Other invertebrates act as predators and filter feeders. Macroinvertebrates are a major food source for most fish species in forested areas (Gregory et al., 1987). Much of the ecological importance of macroinvertebrates stems from their position as an intermediate trophic level between microorganisms and fish (Hynes, 1970).

Benthic macroinvertebrates have several characteristics which make them potentially useful as indicators of water quality. First, many macroinvertebrates have either limited migration patterns or a sessile mode of life, and this makes them well suited for assessing site-specific impacts. Second, their life spans of several months to a few years allow them to be used as indicators of past environmental conditions (Platts et al., 1983). Third, benthic macroinvertebrates are abundant in most streams. Fourth, sampling is relatively easy and inexpensive in terms of time and equipment (EPA, 1989). Finally, the sensitivity of aquatic insects to habitat and water quality changes often make them more effective indicators of stream impairment than chemical measurements (EPA, 1990). In Ohio, for example, 36% of impaired stream segments detected with biosurveys could not be detected using chemical criteria alone (Ohio EPA, 1988).

Disadvantages of monitoring macroinvertebrates include a relatively high degree of variability within or between sites (Minshall and Andrews, 1973), local or regional variations in the sensitivity of a given organism to stress (Winget and Mangum, 1979), the need for specialized taxonomic expertise, and the cost of processing (sorting and identifying invertebrates) samples containing numerous organisms. Much of the variability between samples is due to the highly heterogeneous distribution of macroinvertebrates with depth, current speed, and substratum (Platts et al., 1983; Morin, 1985). This means that sampling locations must be carefully selected and that sampling usually should be stratified by habitat type (Part I, Section 3.3).

Most monitoring techniques require macroinvertebrate identification to genus or species. Interpretation of the results requires knowledge of the habitat requirements of the identified taxa and familiarity with the typical macroinvertebrate community in the study area. In some sampling techniques, considerable effort may be needed to separate organisms from the substrate. The difficulties associated with the separation, identification, and enumeration of taxa may produce inadequate sampling programs (Jackson and Resh, 1988).

Response to Management Activities

The effects of forest activities on macroinvertebrate communities vary. Increases in the riparian canopy opening or the amount of organic material in the streams generally enhance aquatic insect populations. An increase in fine sediment usually has the opposite effect (Gregory et al., 1987; Section 3.1). Removing the riparian canopy decreases the input of terrestrial organic material and the number of detritivores. However, this decline often is overwhelmed by the corresponding increase in primary production and herbivorous insects (Gregory et al., 1987). Several studies have documented an increase in primary productivity after partial or complete removal of the riparian canopy (e.g., Hansmann and Phinney, 1973; Murphy et al., 1981). However, no increase was found in Carnation Creek in coastal British Columbia, where phosphorus was found to be the limiting factor (Stockner and Shortreed, 1988). Logging-induced increases in aquatic insects have been observed in northern California (Erman et al., 1977) and the Oregon Cascades (Murphy et al., 1981). While logging activities may increase total abundance, species diversity is usually reduced (Gregory et al., 1987).

Invertebrate communities also are affected by management practices on forest lands. Buffer strips 30 m wide appeared to protect invertebrate communities from logging-induced changes (Newbold et al., 1980), but buffer strips 10 m wide still resulted in a decrease in detrital inputs and macroinvertebrate densities (Culp, 1988). The net effect of logging on aquatic macroinvertebrates depends on the relative balance among all the controlling factors.

Measurement Concepts

A variety of sampling and data analysis techniques can be used to monitor macroinvertebrate communities. Some of the more common parameters include presence or absence data, functional feeding group analysis, and community parameters. Sample collection techniques can be equally varied, ranging from the placement of uncolonized substrates to kick nets, drift nets, and fixed-area substrate samples.

Sampling Techniques. Sampling techniques for macroinvertebrate can be classified as qualitative, semiquantitative, or quantitative (Platts et al., 1983). Qualitative techniques rely on indicator species or an evaluation of selected functional or taxonomic groups. Generally the samples for qualitative evaluation are not collected on the basis of a specified area or collection effort, and this severely limits any numerical analyses.

Sampling procedures that use uniform substrates or a specified amount of collection effort (e.g., a 3-hour drift net sample, or 50 sweeps with a dip net) are termed semiquantitative techniques (Platts et al., 1983). Data from these samples can be used for qualitative purposes, such as the presence or absence of particular taxa, or for estimating population characteristics such as diversity, total numbers, or biomass. The primary limitation of semiquantitative methods is that results are on a per sample basis rather than per unit area (Platts et al., 1983).

Quantitative techniques involve complete sampling in a specified area. The resulting density data are on an absolute basis (e.g., number of organisms per unit area), and this allows a comparison of populations over time or space. Data collected using quantitative techniques can be used to estimate productivity as well as population characteristics.

Although qualitative techniques typically are quicker and easier than semiquantitative or quantitative procedures, they yield less specific information. This generally makes qualitative techniques less sensitive and less reliable. Since a similar level of expertise is needed to analyze the samples and interpret the results, most projects should use semiquantitative or quantitative sampling methods (Platts et al., 1983).

This range of sampling procedures indicates that a wide variety of sampling techniques have been developed to accommodate varying study objectives and locations. The composition of the substrate, water depth, and current velocity largely determines the most appropriate technique. The most common methods include various types of nets, substrate sampling techniques, and the placement and subsequent retrieval of artificial substrates (Greeson et al., 1977). Each technique has a different set of errors and bias, making comparisons of data from different sampling techniques difficult (Platts et al., 1983). For this reason monitoring studies should select and utilize one of the better-known techniques and apply this as widely as possible to ensure comparable data.

Artificial substrate samplers are useful in large rivers or wherever natural substrates cannot be effectively sampled (EPA, 1989). The most common artificial substrate techniques make use of multiplate (Hester and Dendy, 1962) or basket (Mason et al., 1973) samplers. Multiplate samplers are a set of stacked plates that are left in a stream or lake for a period of at least several weeks and then retrieved for analysis. Basket samplers are similar in principle, but utilize rocks as the substrate for colonization. Advantages and disadvantages of artificial substrates are discussed in Greeson et al. (1977), Rosenberg and Resh (1982), and EPA's Rapid Bioassessment Protocols (EPA, 1989). The most common criticism is that they do not provide a representative sample of the natural community. Major advantages include lower sample variability, and elimination of substrate differences between sample sites.

Drift nets are used to sample macroinvertebrates that have been dislodged or are migrating, and typically they are left in place for at least several hours. However, the nets can become clogged if they are not regularly cleared, and this will reduce the number of organisms captured in the nets. Drift net data are expressed as numbers and biomass of organisms per unit discharge (APHA, 1989).

Dip nets are used to qualitatively collect organisms associated with backwater areas, nearshore areas, and deposits of organic debris. Collection techniques can be specified by area and effort in order to obtain semiquantitative data. In deep waters and in areas with fine substrates, a variety of grab samplers, such as Eckman or Peterson dredges, may prove most effective.

In small forested streams, Surber (Surber, 1937) and modified Hess (Waters and Knapp, 1961; Jacobi, 1978) samplers are most often used for quantitative sampling (Platts et al., 1983). Both of these samplers utilize a frame to delineate a specific area of stream bottom and a net to capture the benthic fauna as the substrate is disturbed to a depth of 5 or 10 cm. The primary difference is that the modified Hess sampler uses a closed frame, while the Surber sampler relies on the current to carry dislodged organisms into the attached net. The mesh size of the net must be large enough to allow the free flow of water and fine sediments, but small enough to capture most of the benthic invertebrates. APHA (1989) suggests a mesh size of 0.595 mm, but in forest streams with little or no fine sediments a smaller mesh size may be preferable. For qualitative or semiquantitative samples, a kick net typically is used. Kick nets can be made by attaching a fine meshed screen between two rods. The net is held vertically in the stream while the substrate immediately upstream is disturbed. The current then carries the dislodged organisms into the net. By specifying the area and effort sampled, semiquantitative data can be obtained (Platts et al., 1983).

Sampling methods must take into account the time of year, number of samples per site, and habitat to be sampled. Significant changes in invertebrate populations occur during the year because of natural life cycle processes (Minshall and Andrews, 1973). To account for these changes, sampling programs must define which season(s) will be sampled and maintain this sampling period throughout the life of the study. Collecting samples in more than one season is preferable, but when this is not possible the optimal sampling season is the period when most macroinvertebrates are both large enough to be retained during sieving and sorting, and identifiable with the most confidence (EPA, 1989). In Region 10 this is typically late winter and early spring. However, sampling effectiveness is reduced during or just after periods of high water. This suggests that the optimal sampling time in streams with snowmelt runoff will be just prior to spring snowmelt, while rain-dominated streams should be sampled after winter storms when the flow regime is relatively stable.

The number of samples that should be collected at each site is a function of the size of the site to be sampled and the and variability between replicate samples. Quantitative methods generally require more samples per site than semiquantitative methods because of the greater variability in invertebrate densities compared to relative abundances (APHA, 1976). In general, quantitative methods will require at least 5-10 samples per site in order to detect statistically significant differences (Platts et al., 1983).

The habitat selected for sampling will greatly affect the type of invertebrate community observed. The most diverse invertebrate communities generally occur in riffle/run habitats with gravel and cobble bottoms (EPA, 1989). Since areas with the greatest diversity will provide the most sensitive indicators to environmental changes, riffle/run habitats are usually preferred for sampling when they are available. Sampling methods developed in North Carolina take qualitative samples from five microhabitats (riffles, macrophytes, logs, sand, and leaf packs) from each site to document invertebrate populations (Lenat, 1988).

Data Analysis. A variety of community and population indices can be used to characterize benthic macroinvertebrates, although the choice will be somewhat constrained by the particular sampling technique used to collect the sample. One useful approach is to divide benthic aquatic insects into functional feeding groups such as shredders, collectors, scrapers, and predators (Cummins, 1973). Changes in the relative abundance of the different functional feeding groups can indicate habitat change. For example, an increase in the number of scrapers as compared to shredders suggests an increase in the production of attached algae due to a reduction in the riparian canopy or an increase in stream width. Considerable care is needed in the separation of organisms, as closely related species can fall into different functional feeding groups. Platts et al. (1983) conclude that this approach shows promise, but still must be regarded as experimental. They recommend that the functional feeding group approach be used in conjunction with more conventional community analysis techniques.

Some of the more commonly used community parameters include abundance, species richness, diversity indices, and biotic indices. Each of these parameters considers only a part of the overall invertebrate population characteristics, and each has certain drawbacks in terms of representing the complex assemblage of organisms present at any given site (Elliott, 1977). It is therefore beneficial to use more than one community measure for assessing invertebrate populations.

Abundance can be expressed in absolute terms as the number of individuals per unit area present, or in relative terms as a percentage of total numbers. The absolute abundance is a useful indicator of the overall productivity at a site. Relative abundance values, such as percent contribution of the dominant taxon, indicate the community balance. Communities dominated by just a few taxa indicate environmental stress (EPA, 1989).

Species richness generally refers to the total number of taxa present. The total number of taxa in specific orders (e.g., total number of mayflies, stoneflies, and caddisflies) also is a useful indicator (EPA, 1989). Lenat (1988) observed a high correlation between species richness and water quality in North Carolina. In Oregon, species richness showed good correlation with trout populations from high

desert streams (Joe Furnish, Vale BLM District, pers. comm.) In some instances, however, moderate degradation may allow new species to colonize a site while not excluding less tolerant species (Gregory et al., 1987). Under these circumstances species richness will be maximized, and a significant decline will not occur until habitat degradation begins to eliminates the less tolerant species. Hence knowledge of the tolerance ranges of different taxa to different pollutants is important for the proper interpretation of species richness data. EPA has published pollution tolerance information on most major aquatic insect orders (e.g., Harris and Lawrence, 1978; Hubbard and Peters, 1978).

Diversity indices combine species richness and relative abundance. A variety of indices have been developed, with the Shannon-Wiener index probably being the most common (Platts et al., 1983). The use of diversity indices for detecting environmental stress has been criticized because they:

1. do not incorporate any trophic community structure,
2. exhibit considerable variation even in undisturbed sites,
3. may be insensitive to disturbance, and
4. are insensitive to the ecological differences between sites (e.g., Pielou, 1975; Zand, 1976).

Various biotic indices have been developed to capture more of the complexities of natural populations. The Biotic Condition Index (BCI) incorporates stream habitat, water quality, and environmental tolerances of aquatic insects (Winget and Mangum, 1979). Tolerances have been estimated or determined for several hundred aquatic insects. The BCI is based on the mean tolerance of the aquatic insects predicted for a site divided by the actual mean tolerances of the aquatic insects found on the site. This method has been used extensively by the Forest Service and the Bureau of Land Management in the Western U.S.

In an effort to provide state governments with a cost-effective integrated biological index, EPA developed five Rapid Bioassessment Protocols (RBP) (EPA, 1989). Protocols I, II, and III use benthic macroinvertebrates to assess water quality impairment; protocols IV and V use fish. RBP I relies upon the qualitative abundance of different macroinvertebrate taxa and professional judgment to determine whether water quality is impaired or unimpaired. It was designed as a quick method to screen different sites (EPA, 1989).

Rapid Bioassessment Protocol II (RBP II) is a more intensive and systematic procedure intended to distinguish among three categories of water quality (non-impaired, moderately impaired, and severely impaired). Separate collections of macroinvertebrates are obtained from riffle/run areas and coarse particulate organic matter. To reduce sample processing time, a 100-organism subsample is randomly sorted from the composited riffle/run samples. Each organism in this subsample is classified to the lowest taxonomic unit (order, family or genus) and functionally by feeding group. Larger subsamples (200 or 300 organisms) can be sorted, but they have not been shown to increase the sensitivity of the procedure (EPA, 1989). The macroinvertebrates collected from coarse particulate organic matter are classified as shredders or non-shredders. From these data eight community, population, and functional feeding group parameters are calculated. These are combined to yield a single evaluation of "biotic integrity," and this is compared to the biotic integrity of a comparable, unimpaired site ("reference station") (EPA, 1989). The particular combination and valuation of parameters in RBP II were developed from a single field study in North Carolina (EPA, 1989), although several of the individual parameters have been derived from previous studies.

RBP III, a more detailed protocol for benthic macroinvertebrates, is very similar to RBP II, but requires identification to the genus or species level. The more precise valuation of the eight metrics allows four levels of impairment (severe, moderate, slight, and no impairment) to be distinguished. Again validation is based on a field study in North Carolina and the use of similar procedures in other studies (EPA, 1989).

Before the Rapid Bioassessment Protocols are implemented in EPA's Region 10, further study is recommended to determine if:

1. riffle/run habitats adequately characterize the "biological integrity" of a stream reach and accurately determine impairment,
2. a subsample of 100 macroinvertebrates is sufficient to characterize the riffle/run community,
3. classification to the family level is sufficient in RBP II,
4. the pollution tolerance data developed for species in other areas are applicable to Region 10, and
5. the selection and combination of the possible metrics used to obtain the biotic integrity are relevant and appropriate in all cases.

Once these methodological questions have been answered, the different protocols must be validated in the different ecoregions of the Pacific Northwest and Alaska.

Standards

The principal objectives of the Clean Water Act are "to restore and maintain the chemical, physical and biological integrity of the Nation's waters" (Section 101). Current water quality programs focus on chemical integrity and, to a lesser degree, on physical integrity (EPA, 1990). It is becoming apparent, however, that chemical criteria do not always protect biological integrity, even though the water quality criteria for parameters such as pH and dissolved oxygen are based in part on the sensitivity of aquatic macroinvertebrates (Part I, Section 1.4; EPA, 1986b). The inadequacy of chemical and physical criteria to protect biological integrity is particularly true for nonpoint source

pollution and habitat degradation. To achieve the goals of the Clean Water Act and protect instream biological integrity, EPA is requiring the incorporation of narrative biological criteria into state water quality standards (Part I, Section 1.4; EPA, 1990).

Current Uses

Fifteen states are now developing biological assessment programs to support future development of biological criteria (EPA, 1990). Some state programs use biological monitoring to evaluate stream impairment, but are not developing specific biological standards. Other states are refining sampling and evaluation methods so that biological standards can be implemented in the future.

Four states currently have biological criteria that are used to enforce water quality standards: Arkansas, North Carolina, Maine, and Ohio. In all four states the biological criteria are based, at least in part, on macroinvertebrate community characteristics (EPA, 1990). Ohio has the most comprehensive biological criteria. Biological indices have been developed for fish and macroinvertebrates for each of the five ecoregions within the state. These indices have been successfully incorporated into the State water quality standards.

In Region 10, Oregon has been using macroinvertebrate assessments to determine stream impairment below point source discharges (R. Hafele, Oregon Dep. Environ. Qual., pers. comm.). Biological assessment methods for monitoring impairment due to nonpoint pollution sources are now being developed in Oregon and Idaho (T. Maret, Idaho Dep. Environ. Qual., pers. comm.). In Oregon macroinvertebrate assessments are an important component of the nonpoint source monitoring program, and Washington is studying the use of macroinvertebrates for monitoring and evaluation.

Assessment

Aquatic insects display several characteristics which make them potentially useful for monitoring purposes. They are relatively sensitive to change, abundant in aquatic ecosystems, and can be directly linked to an important designated use (fisheries). Their use in monitoring has been limited by the difficulties in defining appropriate parameters to measure, the level of expertise required to analyze macroinvertebrate collections, and the difficulty in obtaining representative samples.

The Rapid Bioassessment Protocols (EPA, 1989) are an important step towards establishing sampling procedures and measurement parameters for assessing water quality using macroinvertebrates. Additional work will be needed to establish and verify these assessment procedures for the different ecoregions. Currently the applicability and reliability of the methodology is being studied in several watersheds in Oregon and Washington (R. Hafele, Oregon Dep. Environ. Qual., pers. comm.).

An important limitation of the Rapid Bioassessment Protocols is that they were not designed for quantitative water quality monitoring. The original intent was to develop inexpensive screening tools, and the maximum resolution of the current protocols is four qualitative levels of water quality (EPA, 1989). Quantitative field data may allow additional inferences to be made.

In summary, aquatic macroinvertebrate monitoring is a useful tool for evaluating general water quality condition and the extent to which designated uses are impaired or supported. Biological measurements often are less expensive then detailed chemical analyses, as a trained entomologist can use aquatic insect data to infer a great deal about the site under consideration. To be most effective and reliable, however, biological studies need to be integrated into a monitoring plan that includes both physical and chemical evaluations.

7.4 FISH

Definition

Both resident and anadromous fish communities are found in many of the streams and lakes in forested areas in EPA's Region 10. Twelve salmonid species are commonly found in the forested watersheds of the Western U.S., and these generally are regarded as the most valuable sport and commercial species (Everest, 1987). All of the salmonid species have life stages that are directly affected by management activities and natural disturbances in watersheds. For some water quality parameters, salmonid spawning and rearing is the most restrictive designated use.

Fish are a useful surrogate or integrator of a variety of physical and biological factors. Some of the factors necessary to sustain or restore a particular fish population include the following:

1. adequate streamflow (i.e., water depth and habitat space),
2. sufficient spawning habitat,
3. sufficient rearing habitat,
4. appropriate food sources at different life stages, and
5. proper environmental conditions (particularly temperature, dissolved oxygen, and turbidity).

For anadromous fish there must also be an absence of migration barriers.

The use of fish for monitoring presents many parallels to the sections on algae (Section 6.2) and macroinvertebrates (Section 6.3). Monitoring can be based on the presence or absence of particular species, numbers of a particular species, or community parameters such as productivity, density, and diversity (e.g., Hendricks et al., 1980). The conceptual advantages and disadvantages of these different

parameters are briefly discussed in the following sections, as are the specific techniques which pertain to the use of fish for water quality monitoring.

Relation to Designated Uses

Fisheries are a very important designated use in fresh, estuarine and salt waters. Sport and commercial fishing—primarily of salmonid species—are each worth hundreds of millions of dollars. In many rural areas sport and commercial fishing are major components of the local economy. Fish also have important economic, cultural, and subsistence values for many native Americans.

Ecologically fish are important because they represent the higher trophic levels in streams and lakes. Although fish are the primary predators of macroinvertebrates, their role in the food web varies by species and age. At certain times fish are an important food source for terrestrial fauna such as bears, raptors, and raccoons. Because fish are high in the aquatic food web, they can serve as excellent indicators of the overall physical, chemical, and biological condition of streams.

Salmonids and other large species usually have considerable public appeal. A decline in, or loss of, these species will generate considerable adverse public reaction. Spawning areas, fish ladders, and falls with actively jumping fish may be popular public attractions.

Salmonid species generally have the most stringent habitat requirements. Summaries of habitat requirements for different salmonid species can be found in Everest (1987), Everest et al. (1985), Reiser and Bjornn (1979), and fisheries reference books. Most monitoring activities have focused on salmonids because of their economic importance, strict habitat requirements, and the fact that their habitat requirements generally are better known than most other fish species.

Response to Management Activities

Concern over the response of resident and anadromous fish populations to forest management activities has been a major stimulus to long-term, detailed studies on the effects of forest management on streams. Studies in coastal Oregon (Hall et al., 1987), southwestern British Columbia (Hartman et al., 1987; Chamberlin, 1988), southeastern Alaska (Gibbons et al., 1987), and the Olympic Peninsula in Washington (Cederholm and Reid, 1987) investigated the response of the fish populations to different types and intensities of forest management practices. An important, unifying result of these studies is that forest management can affect a wide variety of physical and biological parameters, including temperature, bed material, primary productivity, peak runoff, low flows, and macroinvertebrate populations. Each of these changes will in turn have a series of effects on fish reproduction, rearing, and growth. The magnitude of these effects will vary by species and age class. In some cases adverse effects on one species may benefit another species.

The complexities of these interacting physical and biological effects makes it very difficult to predict the effects of forest harvest or other management activities. The adoption of increasingly stringent BMPs and forest harvest regulations, particularly in the riparian zone, means that the simple characterizations applied in the past may no longer be appropriate. The detailed, long-term forestry-fisheries studies cited above have demonstrated the need to evaluate impacts by species and life cycle stage, and not rely on single, broad measures such as the total number of fish (e.g., Hartman et al., 1987).

Most of the links between management activities, physical and biological change, and effects on fish are discussed in the context of the individual monitoring parameters such as temperature (Section 2.1), turbidity (Section 4.2), bed material (Section 5.6), and large woody debris (Section 5.7). An excellent review of forestry-fisheries interactions can be found in Salo and Cundy (1987).

Measurement Concepts

A wide variety of techniques have been used to assess changes in the number and condition of fish. In many cases the links between the fisheries measurements, water quality, and management actions are tenuous. Since a complete review is beyond the scope of this document, only the most common and appropriate monitoring techniques are discussed in this section. Edited volumes by Alabaster (1977) and Hocutt and Stauffer (1980) provide good overviews of biomonitoring, while many of the fisheries techniques are discussed in Nielsen and Johnson (1983).

Fish population counts or estimates probably are the most common parameter. For anadromous fish, counts are most often made of the number of fish returning to spawn or the number of fish carcasses following spawning. One also can count the number of outmigrating juveniles (e.g., smolts) from a particular stream or river, but this requires the use and regular maintenance of traps, nets, or weirs. Species which rear for many months in streams, such as coho, are much easier to count than species which outmigrate after emergence and rear in estuaries, such as chum or pink salmon. Counts of outmigrating young provide a more specific indication of spawning and rearing habitat productivity than counts of resident fish or returning adults.

Transient or resident populations within a stream reach can be counted by a variety of means (Platts et al., 1983). Electrofishing is the most common field technique (EPA, 1989), and this is discussed in detail by Reynolds (1983). Electrofishing has the advantage of being relatively accurate and efficient. It is particularly useful in areas which are turbid or have numerous obstructions such as aquatic vegetation, woody debris, or undercut banks. Voltage, pulse, and frequency adjustments are necessary for the following:

1. to reduce size selectivity,
2. to ensure efficient sampling in different-sized streams with varying water quality, and
3. to minimize fish mortality.

The accuracy of population estimates can be improved by making multiple passes with the electroshocker and removing the shocked fish after each pass. Some species can be grouped together for total population estimates, while other species with a different probability of capture must be estimated separately (Platts et al., 1983). Electrofishing allows for the collection of length and weight data, and this can be used to evaluate condition and population structure.

Other methods to capture fish and estimate population size include toxicants and explosives (Platts et al., 1983). While these may allow more accurate population estimates, they kill or alter the populations being counted and now are rarely used.

Direct observation by snorkeling is an increasingly common technique. It is particularly useful in streams with low conductivity and in remote areas. Again there will be variation in the accuracy of the technique by species. Trout and salmon are more likely to hold their territory and be counted, whereas darters and sculpins tend to be more secretive during the day (Platts et al., 1983). Snorkeling can be combined with habitat surveys to provide estimates of species density and species composition for different habitat types (Hankin and Reeves, 1988). Population estimates obtained through snorkeling can be improved by electrofishing in a subsample of the snorkeled habitats (Hankin and Reeves, 1988). In small or steep-gradient streams, direct observations may be limited to pools and glides. The difficulty of obtaining accurate underwater counts means that most surveys provide only an index of the true population. Thus comparisons over time and space can be made only when the counting procedures and conditions are comparable. In general, snorkeling permits a true estimate of fish populations only for certain species under particularly favorable conditions using a carefully executed survey (Platts et al., 1983).

For anadromous species, accurate counts of the returning adult fish and departing smolts can be obtained by placing nets or weir traps on the stream of interest. These capture all migrating fish, but complete counts may require several months. To prevent mortality the captured fish must be regularly removed, and individuals often are counted and weighed at this time. Conlin and Tutty (1979) provide a useful field guide to trapping juvenile salmonids.

An estimate of the number of spawning salmonid pairs can be obtained by counting spawning nests (redds). Ground-based counts are usually more accurate and less costly, and they are the only appropriate technique for smaller forested streams. Aerial surveys may be preferable on larger rivers, but these are usually less accurate (Bevan, 1961). The timing of the redd count is critical because early counts may exclude late-spawning fish, while late counts may underestimate redd numbers because of the decreasing ability to distinguish contemporary redds over time. Redd counts are much more difficult for species or runs that spawn in lakes (e.g., sockeye), large rivers (e.g., chinook), or glacial streams (e.g., spring chinook and sturgeon).

Emergence traps are used to estimate the number of juveniles emerging from a single redd. Emergence success or percent survival through emergence is estimated from an assumed egg count for each species. Low emergence numbers are most often ascribed to infiltrating sediment, but other causes, such as temperature, disease, and predation, must also be considered. Another approach being used in Idaho on a trial basis is to place egg baskets with known numbers of eyed eggs in artificial redds, and use emergence traps to obtain percent emergence.

Species presence or absence, species richness, and diversity indices all have been used as relative or qualitative indicators of water quality (e.g., Warren, 1971; Cairns et al., 1973; Langford and Howells, 1977). The limitations of these parameters have already been discussed (Sections 6.2 and 6.3) and are briefly reviewed for fish in lotic environments in Hendricks et al. (1980). In evaluating these measurements consideration must be given to the biogeographic region, season of measurement, and stream size (Karr, 1981). Generally fewer species of fish occur in undisturbed streams and lakes in the Pacific Northwest than in the Midwest or Southeast, and this hampers the use of diversity or richness measures as indicators of water quality. However, the number of native species may be a sensitive measure of the deterioration of pools and other habitat types (Miller et al., 1988).

Over the last decade there have been several attempts to develop more comprehensive and meaningful measures of fish communities. The index of well being (IWB) incorporates two diversity and two abundance estimates with approximately equal weight (Gammon, 1980). The index of biotic integrity (IBI) is obtained by weighting and summing 12 individual measures (metrics) (Karr, 1981). The metrics were selected on the basis of experience in the Midwest, and they include parameters such as the total number of species, the number of species tolerant and intolerant of poor water quality, several trophic measures, and several indicators of condition (Karr, 1981; Angermeier and Karr, 1986). While this has been widely adopted in the East and Midwest, substantial alterations must be made in order to apply it to sites in Washington, Oregon, Idaho, and Alaska (Hughes and Gammon, 1987; Miller et al., 1988).

The IBI is the basis for EPA's Rapid Bioassessment Protocol (RBP) V. As with RBP II and RBP III (Section 7.3), the habitat quality and IBI for the site under study is compared to the habitat quality and IBI for an unimpaired reference station. Concurrent collection of water quality data is also recommended (EPA, 1989). RBP V is designed to distinguish five levels of water quality impairment (EPA, 1989).

EPA's rapid bioassessment techniques also include a protocol, RBP IV, for quickly assessing the general condition and trend of a particular stream reach. The assessment is based on the completion of a questionnaire by a qualified fish biologist familiar with the stream reach under study. Although data from the questionnaire is qualitative, it is one way to identify reaches needing further study (EPA, 1989).

Standards

At present there are no specific standards or criteria for fish populations or community parameters. However, fish do represent an important designated use in many streams and lakes, and the broad objective of point and nonpoint source water pollution control programs is to protect all designated uses. Hence there is a general standard to protect and maintain natural populations of fish in unimpaired streams and to restore fish communities in streams adversely affected by management. Over the next several years, these general standards will be formalized, as EPA is requiring the addition of narrative biological criteria to state water quality standards. The continuing application and refinement of narrative criteria is intended to lead to quantitative biological criteria within a few years (EPA, 1990).

Recently several species of salmonid fishes have been proposed for listing as threatened or endangered by the U.S. National Marine Fisheries Service. Under the Endangered Species Act, a recovery plan must be prepared for each listed species. This recovery plan could require strict habitat protection measures as well as the direct protection of the designated species. If deemed necessary under the recovery plan, land management activities that might adversely affect habitat quality could be precluded or severely curtailed.

Current Uses

Some advantages of using fish for monitoring water quality are as follows:

1. their mobility and relatively long life span allows them to indicate broad-scale and long-term habitat conditions,
2. their higher trophic position means that they can be used as an integrator of changes in the lower trophic levels,
3. they are relatively easy to collect and identify in the field, and
4. the habitat requirements of many species are relatively well known (EPA, 1989).

Disadvantages include the following:

1. the difficulty of obtaining a representative sample or an accurate estimate of the population,
2. the variety of extraneous factors that can affect fish populations during different life history stages (e.g., fishing pressure, predation, disease) (Hellawell, 1977; Hocutt, 1981), and
3. the mobility and limited residence time of anadromous species in freshwater.

The simple presence or absence of a particular fish species may not be a particularly useful monitoring technique unless we know that it utilized the stream in the past. The mobility and adaptability of fish can result in a few individuals being found even under extremely adverse conditions. For example, the Mt. St. Helens eruption caused the upper part of the Toutle River in Western Washington to have lethal summer temperatures, unsuitable spawning gravels, virtually no cover, highly turbid water, and high winter flushing flows. Yet a few anadromous fish were able to spawn and produce offspring that successfully completed their freshwater stage. Such examples suggest that habitat change sufficient to cause complete loss of a species has to occur on such a scale that monitoring becomes merely a confirmation of the obvious.

In many cases a field examination by a fisheries biologist will permit identification of the key habitat variables for the species of interest. By combining this information with the known or expected management impacts, one can develop a series of hypotheses or questions that will point to specific monitoring technique(s).

This implies a need to identify the causes and effects being ascribed to forest activities, and designing the monitoring program accordingly. A carefully documented decline in fish populations, for example, will only provide the information that a particular population is declining; for a remedial management program to be effective, more specific information is required.

In most cases it will be desirable to monitor each link in the postulated cause-and-effect chain. Concern over fine sediment as a limitation on spawning success, for example, indicates the need to do the following:

1. identify fine sediment sources (perhaps only on a qualitative or reconnaissance basis),
2. monitor changes in bed material particle size or embeddedness (Section 4.6), and
3. evaluate spawning success.

Data on each of these three components are needed to establish a cause-and-effect relationship. Data on the streams or lakes of concern also should be compared to data from unimpaired sites.

The IBI shows promise as a technique to evaluate stream condition from a variety of measurements of fish populations, trophic structure, and species composition (Karr, 1981; Miller et al., 1988). Application to the Willamette River in Oregon required modification of both the scoring system and 5 of the 12 metrics. The IBI more closely corresponded to changes in water quality and substrate than the simpler index of well being (Hughes and Gammon, 1987). Further work is needed to determine the most appropriate metrics and scoring systems for the IBI by ecoregion, size of stream, and type of pollution (Hughes and Gammon, 1987). At this time most of the work on biolo-gical assessment and the IBI is focusing on

macroinvertebrates rather than fish, and this presumably is due to the fact that invertebrates are less mobile, more numerous, more diverse, have a shorter life cycle, and are not as subject to extraneous factors such as fishing pressure.

Assessment

In the Pacific Northwest fish represent an important designated use of most waters. Fish populations can be economically and culturally important, and they often have a high public profile. Often the most stringent constraints on water quality stem from the need to protect coldwater fisheries. The relative absence of certain species from a suitable water body can be a quick and important indication of serious impairment. However, the quantitative monitoring of fish populations, although of critical importance for fisheries management, often is of limited or uncertain value for water quality monitoring.

The limited value of fish for monitoring stems from their mobility, multi-year life span, ecological role, and the numerous extraneous factors that can affect their population. High mobility means that it is difficult to obtain an accurate population estimate, and this limits the likelihood of detecting a statistically significant change. Their multi-year life span may be an advantage in that the number of fish in a certain age group or size class integrates past conditions, but it also is a disadvantage because the number of fish may not provide useful data on current conditions. The position of fish at the top of the food web means that they are affected by any fluctuation at other trophic levels, and this may make it difficult to identify the cause of an observed change. Similarly, interspecific competition is often very important, and this may require an entire set of species to be monitored rather than a single population. Predation is particularly important for alevins and juveniles.

Finally, fish populations can be affected by a wide range of factors unrelated to forest activities, and these greatly complicate any postulated links between fish populations and management. Fishing pressure, disease, hatchery releases, flow conditions, and other factors can affect anadromous and resident fish populations. Anadromous fish populations also are a function of growth and survival rates in the ocean. The inability to accurately estimate the marine mortality of a particular run or population makes it very difficult to relate the returning run size to basin-wide water quality.

Given the numerous factors affecting fish populations and our knowledge of the habitat requirements of many of the most important fish species, it often will be most cost-effective to directly monitor selected habitat parameters and then assume that these will affect fish populations. In many cases, however, fish population data will be needed for stock management and other purposes, and the availability of such data must be considered when designing a water quality monitoring program.

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