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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.

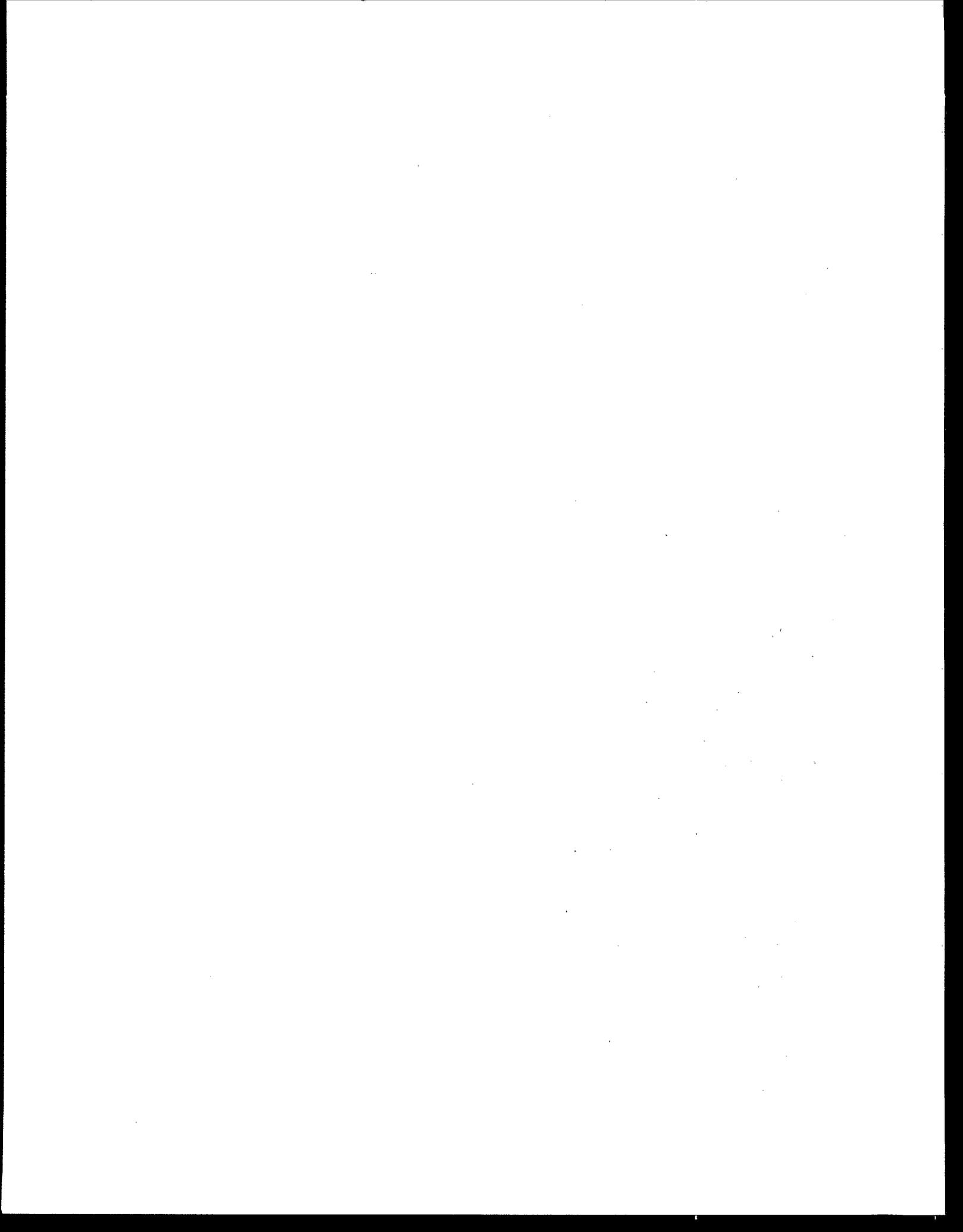
Additional copies of this publication may be obtained from the U.S. Environmental Protection Agency, Region 10, NPS Section, WD-139, 1200 Sixth Ave., Seattle, WA 98101. Copies of the expert system may be obtained by sending a diskette formatted in MS-DOS to the same address.

ABSTRACT

This document provides guidance for designing water quality monitoring projects and selecting monitoring parameters. Although the focus is on forest management and streams in the Pacific Northwest and Alaska, a broader perspective is taken, and much of the information is more widely applicable.

Part I reviews the regulatory mechanisms for nonpoint source pollution and defines seven types of monitoring. A step-by-step process for developing monitoring projects is presented. Because monitoring is a sampling procedure, study design and statistical analysis are explicitly addressed. The selection of monitoring parameters is defined as a function of the designated uses, management activities, sampling frequency, monitoring costs, access, and the physical environment. Approximately 30 parameters are rated with regard to these controlling factors. A qualitative combination of these rankings yields recommended monitoring parameters for various management activities. This parameter selection process has been incorporated into an interactive PC-based expert system called PASSSFA.

Part II is a technical review of the parameters, which are grouped into six categories: physical and chemical constituents, flow, sediment, channel characteristics, riparian, and aquatic organisms. The review of each parameter is organized into seven sub-sections: definition, relation to designated uses, response to management activities, measurement concepts, standards, current uses, and assessment.



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PREFACE

Over the last few decades, the forest industry has come under increasing scrutiny, and this is particularly true in the Pacific Northwest. Since World War II the economy has been moving away from primary industries to service and manufacturing, and there has been a rapid shift to an urban-based population. This, together with a greater awareness of environmental issues, has led to increasing public concern over the adverse effects of forest land management on water quality and stream condition. Much of the public attention has been directed towards the dramatic decline in the number of salmonid fishes in many of the major river basins, and the resultant economic, cultural, and legal implications. Considerable concern also has been expressed over the impact of land management activities on the other designated uses of water, such as domestic water supply and recreation, and the other values of water bodies that may not be recognized, such as the health of aquatic and riparian ecosystems. The current trend is clearly towards increasingly stringent regulation of forest practices, and there are no signs that public concern will abate in the future.

Passage of the National Environmental Policy Act (NEPA) in 1969 provided a means for the regulatory agencies and the public to openly evaluate the potential environmental impacts of major management actions and participate in the federal planning process. However, there often is not a comparable, clearly defined process by which the public and regulatory agencies can evaluate the effects of management activities on the environment. This is particularly true in the forestry arena, as nonpoint source pollution is controlled primarily by the formulation and adoption of Best Management Practices (BMPs). Effective BMP evaluation can be done only by directly monitoring the effects of management activities on the designated uses of the water bodies of concern. Nearly 20 years of experience has shown that the protection of streams through BMPs is an iterative process, and state water quality agencies, together with the U.S. Environmental Protection Agency (EPA), have been given the primary responsibility for overseeing the adoption, implementation, and evaluation of the management practices needed to adequately protect water quality. Clearly this mandate can be carried out only if there is an effective means to monitor the effects of forestry activities on aquatic ecosystems and the designated uses of water.

An ideal parameter for monitoring the impacts of a land management activity such as forestry should

- be highly sensitive (responsive) to the management action(s),
- have low spatial and temporal variability,
- be accurate, precise, and easy to measure, and
- be directly related to the designated uses of the water body.

This ideal parameter should then be monitored in the context of a project which will (1) provide useful feedback to the managers, (2) directly link management activities to the status of the designated uses both on-site and downstream, (3) allow statistical inferences to be made to larger populations, and (4) allow quantitative estimates of risk and uncertainty. Since such ideal parameters do not exist and monitoring projects rarely are able to fulfill all these objectives, Region 10 of the U.S. EPA, which includes Washington, Oregon, Idaho, and Alaska, proposed a year-long project to develop guidelines for monitoring the effects of forestry-related activities on streams. The present document stems from that initial proposal, and it addresses both the design of monitoring projects and the selection of parameters for monitoring nonpoint sources of pollution in forested areas.

These *Guidelines* follow a tradition of concern at EPA for the aquatic effects of management activities in forested areas. Earlier publications addressed road construction (EPA, 1975), timber harvest (EPA, 1976), the application of forest chemicals (EPA, 1977), an evaluation of nonpoint silvicultural sources (EPA, 1980), and effectiveness of nonpoint controls (EPA, 1988). Our hope is that this document will stimulate further analysis and progress in the design and execution of water quality monitoring projects.

The field of water quality monitoring in forested areas is still young, and the preparation of these *Guidelines* made apparent the relative paucity of published information on the results of monitoring projects and many of the parameters evaluated in this document. Even "unsuccessful" monitoring projects can help direct future monitoring efforts if the results are properly evaluated and disseminated.

These *Guidelines* represent our best effort to define the key elements that lead to a successful monitoring project. By direct interviews, literature reviews, and the generous assistance of numerous experts, we have attempted to summarize the state of the art and anticipate future developments. We fully recognize that monitoring nonpoint source pollution in forested areas is, like management, an iterative process. We hope that these *Guidelines*

will provide the basis for continued improvements, as the future of our streams in forested areas will largely depend upon the quality of our monitoring efforts.

EPA, 1975. Logging road and protection of water quality. U.S. Environmental Protection Agency, Region 10, Water Division, EPA 910/9-75-007. Seattle, WA. 312 p.

EPA, 1976. Forest harvest, residue treatment, reforestation and protection of water quality. U.S. Environmental Protection Agency, Region 10, Water Division, EPA 910/9-76-020. Seattle, WA. 273 p.

EPA, 1977. Silvicultural chemicals and protection of water quality. U.S. Environmental Protection Agency, Region 10, Water Division, EPA 910/9-77-036. Seattle, WA. 224 p.

EPA, 1980. An approach to water resources evaluation of non-point silvicultural sources (A procedural handbook). U.S. Environmental Protection Agency, Environ. Res. Lab., EPA-600/8-80-012. Athens, GA. 836 p.

EPA, 1988. Effectiveness of agricultural and silvicultural nonpoint source controls. U.S. Environmental Protection Agency, Water Div., Region 10, EPA 910/9-88-210. Seattle, WA. 181 p. + app.

ACKNOWLEDGMENTS

The development and execution of a project such as this draws heavily upon the time and effort of a large number of people. Generally projects are most vulnerable during the early stages of formulation, and in this case most of the initial impetus came out of the Region 10 office of the U.S. Environmental Protection Agency (EPA) in Seattle. Tom Wilson, Chief, Office of Water Planning, and Elbert Moore, Nonpoint Source Coordinator, both played key roles in developing the conceptual basis for the project and garnering support. Their shepherding role continued as the project was funded and brought to completion, and we deeply appreciate their efforts and especially their patience.

A Project Steering Committee was formed at the Center for Streamside Studies (CSS) at the University of Washington to oversee the formulation of a workplan, the hiring of a coordinator, and to assist with the preparation of the document. Members of the Project Steering Committee included Dr. Robert Naiman, Director of CSS; Dr. Ken Raedeke, CSS; Dr. Dennis Harr, a research scientist for the U.S. Forest Service based at the University of Washington; Dr. Loveday Conquest, Center for Quantitative Studies at the University of Washington; Mr. Steve Ralph, Coordinator for the Timber/Fisheries/Wildlife (TFW) Ambient Monitoring Program based at CSS; and the three authors of the *Guidelines*. The assistance of Drs. Harr and Conquest in reviewing earlier drafts of selected sections of the *Guidelines*, often at short notice, is particularly appreciated. The conceptual links between this project and the TFW Ambient Monitoring Program led to a continuous exchange of ideas and experience with Steve Ralph, and his contributions to the project are too numerous to list.

A Technical Advisory Committee was convened to review the *Guidelines* at two different stages. Each member gave generously of their time to attend these meetings and provide written comments on early drafts, and their inputs were critical to the preparation of this document. Members included Dr. Robert Beschta, Oregon State University; Dr. Robert Bilby, Weyerhaeuser Co.; Mr. Bill Brookes, Bureau of Land Management; Mr. Tim Burton, Idaho Department of Health and Welfare; Dr. C. Jeff Cederholm, Washington Department of Natural Resources; Mr. Jere Christner, Tongass National Forest; Mr. Jim Doyle, Mt. Baker-Snoqualmie National Forest; Mr. Rick Hafele, Oregon Department of Environmental Quality; Dr. George Ice, National Council of the Pulp and Paper Industry for Air and Stream Improvement; Dr. K Koski, U.S. Fish and Wildlife Service; Dr. Dale McCullough, Columbia River Intertribal Fish Commission; Mr. Stuart McKenzie, U.S. Geological Survey; Dr. Walter Megahan, U.S. Forest Service Intermountain Research Station; Mr. John Potyondy, Boise National Forest; Dr. Gordon Reeves, U.S. Forest Service Pacific Northwest Research Station; Mr. Jim Ryan, Washington Department of Natural Resources; Mr. Jack Skille, Idaho Department of Health and Welfare; Mr. Rick Stowell, Nez Perce National Forest; Mr. Jon Vanderheyden, Wallowa-Whitman National Forest; Mr. Dick Wallace, Washington Department of Ecology; and Dr. Larry Wasserman, Yakima Indian Nation. Several members of the Technical Advisory Committee reviewed a final draft of the entire document, and their efforts to do so over a relatively short time period, despite their other commitments, are most appreciated.

A number of people outside of the Technical Advisory Committee helped review and revise sections in which they had particular expertise, and we would like to acknowledge the contributions of Mr. Pete Bisson, Weyerhaeuser Company; Dr. Bill Dietrich, University of California at Berkeley; Dr. Gordon Grant, U.S. Forest Service Pacific Northwest Experiment Station; Dr. Stan Gregory, Oregon State University; Dr. John Stednick, Colorado State University; and Mr. Gerald Swank and Mr. Glenn McDonald, both at the Forest Service Pacific Northwest Regional Office. Mr. Gino Luchetti, King County, Washington, and Mr. Bruce McCammon, Mt. Hood National Forest, each contributed ideas and a case study. Additional individuals within EPA, other government agencies, and several universities were called upon for their comments, and we hope that these individuals will see that their suggestions have helped to strengthen the *Guidelines*.

A final set of thanks must go to graphics designer Ms. April Richardson and the Research Publications Editor at the School of Fisheries, Mr. Marcus Duke. April combined our various and diverse ideas into a coherent cover design, while Marcus transformed rough text and mere sketches of figures and tables into final form. Without Mr. Duke's meticulous help and patience, this document might never have become a finished publication.

Despite the contributions of all these individuals to the *Guidelines*, final responsibility for its content and remaining shortcomings rests with us. We emphasize that in preparing this document we viewed it not as an ultimate truth, but as a set of guidelines which must be evaluated and applied with discretion. It is our hope that the *Guidelines* will prove useful to a wide audience. However, even if the *Guidelines* do no more than stimulate further thought and a more critical design and execution of water quality monitoring projects, we will have achieved a major portion of our objectives.

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GLOSSARY

<	less than
>	greater than
bankfull stage	the water surface elevation of a stream flowing at channel capacity
bankfull discharge	discharge at bankfull stage
BMP	Best Management Practice
BOD	biological oxygen demand
cm	centimeter
CSS	Center for Streamside Studies
dbh	diameter breast height
DO	dissolved oxygen
EPA	U.S. Environmental Protection Agency
ft	foot/feet
g	gram
ha	hectare
hr	hour
IBI	index of biotic integrity
ISI	interstitial space index
IWB	index of well being
L	liter
LC-50	lethal concentration, 50%; the concentration of a toxin or pollutant that kills half of the organisms in a test population per unit time
LWD	large woody debris
m	meter
μ	micro (10 ⁻⁶)
mg	milligram
ml	milliliter
mm	millimeter
min	minute
NPDES	National Pollutant Discharge Elimination System
ppb	parts per billion
ppm	parts per million
PCE	percent cobble embeddness
RAPID	Rapid Aerial Photographic Inventory of Disturbance
RBP	Rapid Bioassessment Protocol
reach	a continuous portion of a stream between two designated points
sec	second
TMDL	total maximum daily load

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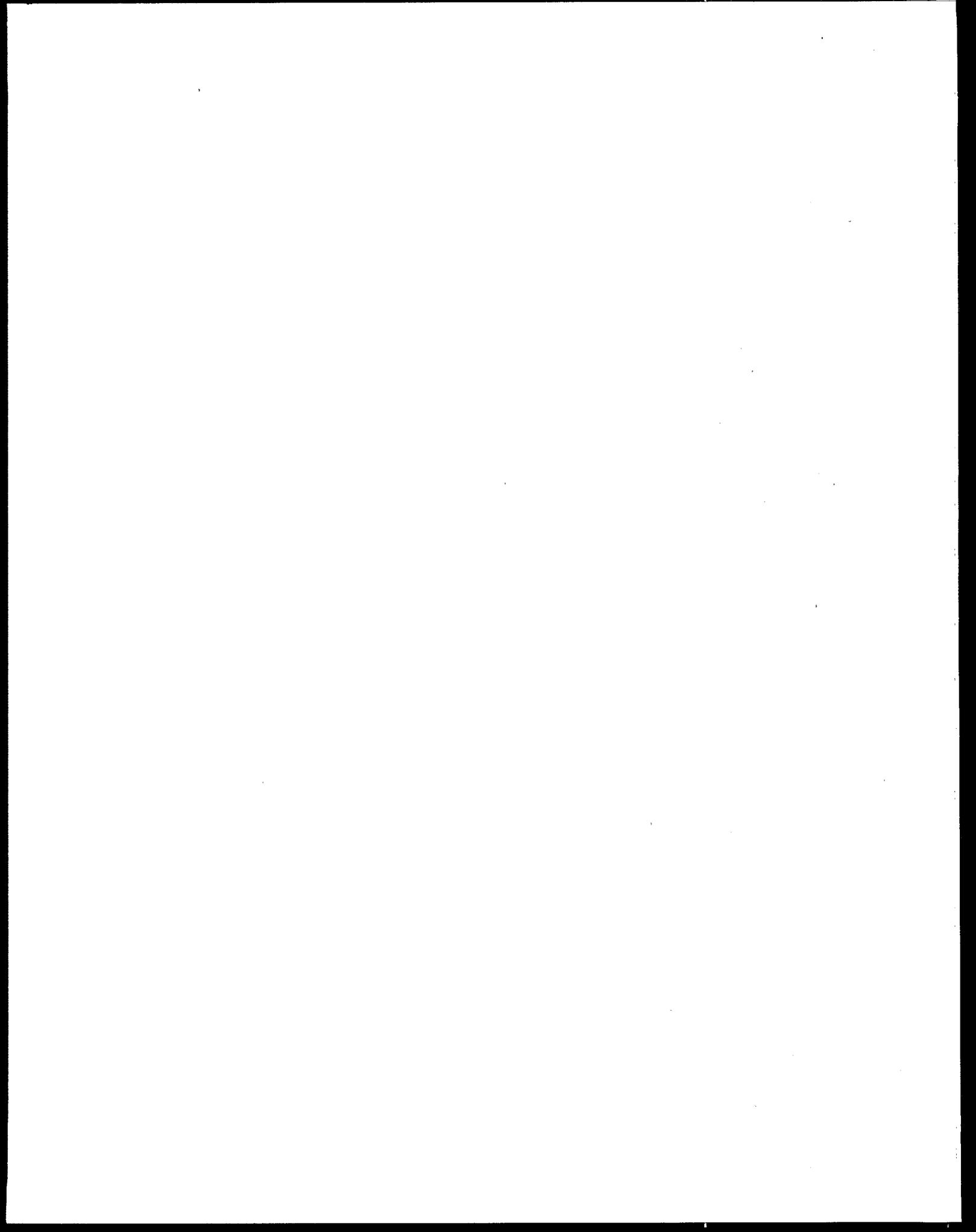
EXECUTIVE SUMMARY

1. The purpose of these *Guidelines* is to assist people in developing water quality monitoring plans. This includes the design of monitoring projects and the selection of monitoring parameters. The rationale for the *Guidelines* is that nonpoint sources of pollution more commonly limit the designated uses of water in forested areas than point sources. Effective monitoring projects are essential to determine trends, evaluate control efforts, and assess the impact of management activities on the designated uses of water.
2. The scope of the *Guidelines* is limited to forested areas in Washington, Oregon, Idaho, and Alaska (Region 10 of the U.S. Environmental Protection Agency). This helps reduce the variability in the range of conditions to be monitored, and the number of management activities that must be evaluated. Although the focus is on the effects of forestry and forestry-related activities on streams, other management activities that often occur in forested areas (e.g., grazing, mining, and recreation) also are discussed because they directly affect water quality in forested areas, and the effects of these other activities generally cannot be monitored independently from forest management activities. Similarly, the *Guidelines* focus on streams and do not directly address monitoring procedures in lakes, reservoirs, and other downstream areas. However, the *Guidelines* explicitly recognize that upstream changes in water quality can affect downstream designated uses, and this must be considered when formulating a monitoring plan and selecting monitoring parameters.
3. For these *Guidelines* water quality is defined in the broadest possible sense. Hence the monitoring parameters include not only the traditional physical and chemical constituents of water, but also those parameters which directly affect the designated uses of water. A total of 30 parameters or groups of parameters are evaluated and reviewed, and these are as follows: physical and chemical constituents (temperature, pH, conductivity, dissolved oxygen, nitrogen, phosphorus, herbicides and pesticides); flow characteristics (size of peak flows, amount of low flow, water yield); sediment (suspended sediment, turbidity, bedload); channel characteristics (cross-section, width and width/depth ratio, pool parameters, thalweg profile, habitat units, bed material particle size, embeddedness, surface vs. subsurface bed material particle size, large woody debris, bank stability); riparian characteristics (riparian canopy opening, riparian vegetation); and aquatic organisms (bacteria, algae, macroinvertebrates, and fish).
4. Seven types of monitoring are defined—trend, baseline, implementation, effectiveness, project, validation, and compliance monitoring. Because of the focus on instream, channel, and riparian parameters, the *Guidelines* generally are less applicable to implementation monitoring and some types of effectiveness monitoring.
5. The legal background for water quality monitoring is reviewed. Two key roles for water quality monitoring are to determine if the designated uses for a particular water body are being impaired, and whether water quality standards are being met. Answers to these questions often determine the type and intensity of monitoring activities. Regular feedback of the monitoring results through well-defined feedback loops is an essential component of any monitoring project. The design and execution of monitoring projects must be considered an iterative process, as the process of data collection and analysis inevitably will have implications for the frequency, location, and type of measurements.
6. The most important step in developing a monitoring plan is to clearly define the objectives. A clear and detailed set of objectives will help preclude unrealistic expectations and greatly facilitate the design of a monitoring plan. A pilot project can prove extremely useful and cost-effective when there is some uncertainty about the type and location of monitoring activities.
7. The statistical considerations of water quality monitoring are very important since water quality monitoring is a process of sampling selected parameters in space and over time. Replication of samples, treatments, and controls is essential if any statistical inferences or generalizations are to be made. The use of statistics permits quantitative estimates of risk, error, and uncertainty.
8. Common designs and sampling procedures are briefly reviewed with regard to water quality monitoring. Examples and case studies are used to illustrate some of the salient points. Key problems discussed include overlapping management activities, determination of cause-and-effect, separation of natural and anthropogenic causes, and the potential time lag between management activities and changes in the parameter being monitored. The trade-offs between sample size, sample variability, level of significance, power (probability of detecting a real difference), and minimum detectable effect are explicitly reviewed and illustrated.

9. The selection of monitoring parameters is presented as a function of the designated uses of the water body being monitored, the type of management activities, and the cost of monitoring. Monitoring costs are broken into the frequency of sampling, the range of flow conditions needing sampling, data collection time, equipment costs, and analytic costs. All 30 parameters evaluated in the *Guidelines* are qualitatively ranked with regard to each controlling factor in a series of tables. Access to the proposed monitoring sites, the availability of existing data, and the physical environment (e.g., climate, land form, geology, and soils) are other important factors that cannot be qualitatively ranked, but which influence the relative value of the different monitoring parameters.
10. Chapter 5 of Part I integrates these qualitative rankings to provide an overall evaluation of the usefulness of each parameter for the 10 different management activities considered in the *Guidelines*. The rationale for the relative rankings presented in Table 5 is briefly discussed for each management activity.
11. The results indicate that the choice of monitoring parameters is rarely clear. For monitoring forest management activities, most of the traditional physical and chemical parameters have only limited usefulness because of their relative insensitivity, their high cost of monitoring, or both. The parameters related to channel characteristics are promising because of their relative sensitivity and low measurement costs. Aquatic organisms—particularly invertebrates—also have some specific features that may prove useful for monitoring, but more work is needed before they can be widely utilized in the Pacific Northwest and Alaska.
12. A limitation of the procedure for selecting the most useful parameters is that each parameter is considered independently. A final table evaluates the magnitude of the interrelationships between each possible pair of parameters, and this helps to identify those parameters that may be overlapping or redundant with regard to monitoring particular management activities.
13. The second part of the *Guidelines* is a technical review of each of the parameters evaluated in Part I, and this is designed to provide an overview of each parameter and to serve as a reference section. For each parameter there are seven sub-sections: (1) definition, (2) relation to designated uses (i.e., how changes in the parameter affect the designated uses of water), (3) effect of management activities on the parameter, (4) measurement concepts, (5) standards, (6) current uses, and (7) assessment. The assessment section is designed as an overall, qualitative summary of the parameter as it relates to water quality monitoring—particularly for forestry activities—in streams in the Pacific Northwest and Alaska.
14. The technical review of each parameter provides a summary of the relevant literature, but it is not a comprehensive review or an operational manual. However, the numerous literature citations allow rapid identification of sources for more detailed information, including field measurement techniques and analytic procedures.
15. The parameter selection procedure presented in Part I has been incorporated into a PC-based expert system called PASSSFA (Parameter Selection System for Streams in Forested Areas). The executable version of the expert system allows users to quickly identify appropriate monitoring parameters through an interactive series of questions and answers. The confidence level assigned to each recommended parameter provides a relative indication of the likely usefulness of that parameter given a particular set of management activities, designated uses, and monitoring constraints. A “what if” function allows the user to quickly alter his or her response to a particular question and then generate a revised list of recommended monitoring parameters.

PART I

DEVELOPING A MONITORING PROJECT



1. INTRODUCTION

1.1 PURPOSE OF THE *GUIDELINES*

The purpose of this document is to assist land use managers and their technical staff in developing water quality monitoring plans for forested areas in Washington, Oregon, Idaho, and Alaska. The document focuses on the design of monitoring projects and the selection of the parameters to be monitored given (1) the designated and existing water uses, (2) the type and intensity of management activities, (3) the environmental setting, and (4) the monitoring objectives and constraints. Even though the discussion and examples are directed towards forest management activities in the Pacific Northwest and Alaska, at least the monitoring principles and the hierarchy of decision-making should be broadly applicable both to other management activities and to non-forested environments. The *Guidelines*—particularly the technical reviews of individual parameters in Part II—are not intended to provide a step-by-step guide to field procedures and analytic techniques, as this information is readily available from the references cited in the text.

The rationale for developing these *Guidelines* stems from the increasing emphasis on controlling nonpoint sources of water pollution, and the recognition that monitoring is an essential component of any water pollution control program. The emphasis on nonpoint sources is due to the realization that nonpoint sources are the major cause of water quality impairment in rivers and lakes in the U.S. (EPA, 1989a). Better control of nonpoint sources is necessary if the broad objectives of the Clean Water Act are to be achieved.

The conceptual and methodological problems associated with assessing and monitoring nonpoint sources are quite distinct from those associated with point sources. For point sources the quantity and type of pollution usually can be measured prior to its release. Comparable data cannot easily be collected for nonpoint sources, so the assessment and monitoring of nonpoint source pollution must rely on

data collected in the receiving waters. This greatly complicates monitoring, as the pollution is diluted and it may be difficult to separate management effects from natural processes. Furthermore, many of the parameters and measurement techniques used to characterize point source effects cannot be applied to nonpoint sources. These difficulties are particularly prominent for forest management activities, as the resultant pollution often represents a change in an existing value rather than the introduction of entirely new pollutants. Hence the two primary objectives of this document are (1) to provide guidelines for developing effective nonpoint source monitoring plans, and (2) to review the parameters that are or might be useful for monitoring nonpoint sources of pollution in forested areas.

Throughout this document the term water quality is used in the broadest possible sense. This means that water quality includes not only the traditional physical and chemical constituents such as pH, temperature, and discharge, but also those parameters that affect the existing and designated uses of a water body. In many parts of the Pacific Northwest, for example, maintenance of salmonid fisheries is an important designated use. The need to protect this use necessitates concern over other water quality parameters such as the amount of large woody debris, the number and size of pools, the density of the riparian canopy, and the particle size of the bed material. Although these are not normally included in water quality monitoring projects, each of these parameters is relatively sensitive to certain forest management activities and directly related to habitat quality (Section 4.3). We expect that future programs to evaluate and control nonpoint source pollution will have to explicitly acknowledge this broader definition of water quality, and that some of these parameters may prove more useful or important than the traditional chemical and physical constituents of water quality. Hence these *Guidelines* evaluate and review nearly thirty different parameters pertaining to the major designated uses of water in the Pacific Northwest and Alaska.

Part I

Many of the parameters, such as habitat types and aquatic macroinvertebrates, actually incorporate a large number of specific measurements, but were grouped as a single parameter for practical reasons.

The restriction of this document to nonpoint source pollution in forested areas means that the management activities of primary concern are forest harvest, road construction and maintenance, forest fertilization, pest and weed control, and recreation. Other aspects of forest management, such as mechanical site preparation and intermediate stand treatments, are not explicitly considered, as their impact on water quality is conceptually similar to the impact of forest harvest and road building, and typically lesser in magnitude.

Grazing, concentrated recreation, mining, and rural settlements are other management activities that commonly occur in forested areas and which can adversely affect water quality. Although these other management activities are outside the scope of this document, it is often inefficient or impossible to monitor only the impact of forestry activities, and not simultaneously consider the effects of mining, grazing, recreational developments, or fire. Hence the selection of parameters to monitor each of these other management activities is briefly discussed in Chapters 4 and 5. To the extent possible, the parameter reviews in Part II also acknowledge the importance of these other management activities. Relatively little attention is devoted to the effects of rural communities and recreational developments on water quality, as extensive literature already exists on this topic. Mining impacts also are not discussed in detail because they are so variable with regard to the minerals being extracted, the method of operation, and the processing techniques. In general the review of specific parameters in Part II should prove helpful in developing monitoring projects that explicitly consider and evaluate one or more of these other management activities.

The *Guidelines* are designed to be most applicable to perennial streams and small rivers. In larger river systems water quality usually is controlled by agricultural, industrial, and municipal wastes, and it becomes very difficult to distinguish the impact of forest management activities. Furthermore, some of the parameters discussed in the *Guidelines* (e.g., habitat types) cannot be easily applied to large rivers.

The *Guidelines* also do not address the design of water quality monitoring projects for lakes, as lakes are so distinct in terms of their physical and biological characteristics. Most lake monitoring projects rely on the collection and analysis of water samples, while a much broader range of monitoring parameters can be used in streams and rivers. Although some of the parameters discussed in the *Guidelines* are relevant to water quality monitoring in lakes, the *Guidelines* do not explicitly address their potential use in lacustrine environments.

However, the *Guidelines* explicitly consider the need to protect water quality in downstream lakes and reservoirs, as

this can be an important designated use of water from forested areas. Concern over downstream lake water quality may affect the design of a stream monitoring project by requiring additional parameters to be monitored, or by further constraining the allowable change in certain parameters. The *Guidelines* explicitly identify such situations, and suggest which parameters are most appropriate for protecting lake water quality.

Finally, the *Guidelines* consider only those measurements which can be made either in or immediately adjacent to the stream channel. Observations on upslope areas often are essential to understanding the cause of changes observed in the stream channel, and they also may exhibit a higher sensitivity to management actions than inchannel measurements. Nevertheless, upslope measurements represent a completely different set of monitoring techniques that are not addressed in the present document.

1.2 ORGANIZATION AND USE OF THE GUIDELINES

The *Guidelines* are divided into two parts. Part I presents the background and principles of developing a water quality monitoring plan for nonpoint source pollution in forested areas. It includes a discussion of the factors that should be considered in developing a monitoring plan, and a set of tables summarizing the sensitivity, cost, and usefulness of the various monitoring parameters vis-a-vis the designated uses of water and the management activity to be monitored. Part II presents individual technical reviews of the monitoring parameters that have been or might be used to monitor water quality in forested areas.

A user's guide to this document is provided in Box 1. This indicates that those who wish only to learn more about a certain parameter should go directly to the relevant section in Part II.

Those who have a clearly defined monitoring objective, but are uncertain about the parameters to be monitored, can use the five tables in Chapters 4 and 5 of Part I as a qualitative guide to the most appropriate parameter(s). Table 2 (page 39) rates each parameter with regard to its effects on different designated uses (i.e., how does a change in parameter X limit different designated uses). Table 3 (page 41) rates the sensitivity of the parameters to different management activities (i.e., how likely is parameter X to change as a result of various management activities). Table 4 (page 43) provides a general indication of the "typical" frequency and sampling cost associated with each parameter. Table 5 (page 50) integrates the data from Tables 2-4 to provide a qualitative evaluation of the usefulness of each parameter for monitoring different management activities. Tables 6A and 6B (page 62) summarize the interactions among the parameters; these tables can be used to identify

Box 1. HOW TO USE THIS DOCUMENT

Guidance on specific aspects of developing a monitoring plan can be found as follows:

Defining objectives	Part I, Section 4.1
Identifying types of monitoring	Part I, Section 1.3
Understanding the legal framework for monitoring	Part I, Sections 1.4 and 2.1
Sampling, statistical design, and statistical trade-offs	Part I, Chapter 3
Factors which constrain and help define a monitoring plan	Part I, Chapter 4
Recommended monitoring parameters	Part I, Section 5.1
Parameter interactions	Part I, Section 5.3

A quick and generalized evaluation of the monitoring parameters with regard to the following topics can be found in the table as indicated:

Effect on designated uses	Table 2, p. 39
Sensitivity to management activities	Table 3, p. 41
Frequency and cost of sampling	Table 4, p. 43
Recommended use for monitoring different management activities	Table 5, p. 50-51
Interactions (i.e., the effect of one parameter on another parameter)	Tables 6A and 6B, p. 62-65

Information on the expert system can be found in Part I, Section 5.2.

More information about specific monitoring parameters can be found in the appropriate section of Part II.

Physical and chemical constituents	p. 71
Discharge parameters	p. 90
Sediment parameters	p. 96
Channel characteristics	p. 107
Riparian condition	p. 131
Aquatic organisms	p. 138

closely related parameters that might be redundant or necessary to evaluate the cause of an observed change.

It should be recognized that these tables represent a generalized, qualitative evaluation that will not apply in all cases. The diversity of environments and processes in the Pacific Northwest and Alaska, together with the variation in management, precludes any absolute evaluation. Hence the tables should be regarded as an initial guide, and the values will need to be adjusted according to local knowledge and experience. The tables should also be interpreted in the context of the text in Chapters 4 and 5, as these two chapters outline the principles behind the qualitative rankings and provide more detail on how the rankings might be adjusted under different circumstances.

Sections 1.3 through 3.4 of Part I will be useful to those who are uncertain about the structure and concepts that must be considered when developing a monitoring plan. Again the diversity of objectives, environments, and management activities means that the emphasis in the *Guidelines* is on basic principles, and the user will need to apply these to their particular situation. In Section 1.3 the various types of monitoring are described, while in Section 1.4 the legal background for monitoring nonpoint sources is provided.

Chapter 2 places the types of monitoring into the legal framework described in Section 1.4, and then outlines the basic process of developing water quality monitoring plans.

Chapter 3 discusses the statistical considerations in monitoring. It points out that monitoring is inherently a sampling procedure. This means that questions of statistical design and analysis must be explicitly recognized. Again this chapter does not provide step-by-step procedures, but emphasizes the understanding and application of basic principles.

Chapter 4 reviews the factors that must be considered in developing a monitoring plan. Formulation of specific objectives is seen as the single most important step. The remaining sections of Chapter 4 discuss the other important considerations involved in formulating a monitoring plan. These include the designated uses of water, type of management activity,

frequency and cost of monitoring, availability of existing data, and the physical environment. Although the discussion and examples focus on land management activities in forested areas in the Pacific Northwest and Alaska, the principles and hierarchy can be applied both to other management activities and non-forested environments.

Chapter 5 qualitatively ranks the parameters with regard to their usefulness for monitoring the water quality effects of land use actions in forested areas. These recommendations represent an integration of the information in Chapters 1-4 and in Part II. Again it should be emphasized that these rankings are not absolute, but a relative evaluation of which parameters should be most generally applicable. The second part of Chapter 5 discusses the interactions among the various parameters, and Tables 6A and 6B show how each parameter affects all of the other parameters discussed in the *Guidelines*. This information helps identify those closely related parameters which (1) might be another source of change for the parameters being monitored, or (2) might serve as a surrogate for the parameter of interest.

Part II of this document can be viewed as a reference section for each of the monitoring parameters discussed in Part I and ranked in Tables 2-6. The individual parameters are grouped into five classes as follows: water column

Part I

measurements (temperature, pH, conductivity, dissolved oxygen, nutrients, herbicides, and pesticides); discharge parameters (size of peak flows, low flows, and water yield); sediment parameters (suspended sediment, turbidity, and bedload); channel characteristics (channel cross-section, width and width-depth ratio, pool parameters, thalweg profile, habitat type, bed material size, embeddedness, large woody debris, and bank stability); riparian condition (riparian canopy opening, riparian vegetation); and biological components (bacteria, algae, invertebrates, and fish).

For convenience the discussion of each parameter is divided into seven sub-sections:

1. definition,
2. relation to designated uses,
3. response to management activities,
4. measurement concepts,
5. standards,
6. current uses, and
7. assessment.

The assessment section is a qualitative evaluation and summary, and it can be read separately if desired. The extensive references at the end of Part II direct the reader to more detailed sources of information on each parameter.

In summary, the *Guidelines* are designed to help guide the development of a monitoring plan, with particular emphasis on the selection of the parameters to be monitored. It must be recognized, however, that the *Guidelines* are subject to several limitations. First, the tables are a qualitative evaluation based on a combination of experience and published data. We have tried to integrate the views of many experts, but there will always be some divergence of opinions. Second, it is not possible to develop a set of guidelines which will apply in all environments for all conditions. Any divergence between the *Guidelines* and one's individual views should be used to stimulate further discussion and a critical reassessment. Ultimately, however, local knowledge and experience should take precedence over any generalized guidelines. Third, the discussion and matrices are based on current knowledge. In many cases data on the sensitivity and variability of a parameter are not available, or are known only for a particular environment. As more data are accumulated, our opinion as to relative usefulness of a parameter may change. Finally, measurement techniques are evolving, and this will affect the ease of measurement, the inherent variability, and the sensitivity to detect change.

1.3 TYPES OF MONITORING

The term "monitor" is defined as to watch or check. Although it is not an explicit part of the definition, the term monitoring suggests a series of observations over time. This repetition of measurements over time for the purpose of detecting change distinguishes monitoring from inventory and assessment. While both inventories and assessments

can be based on a single measurement or observation, they also can incorporate a series of observations to obtain a better estimate of a particular parameter. For example, the number of species of fish in a particular reach might be counted as part of an inventory of fish species, and several counts might be made in order to obtain a more accurate estimate. Similarly, maximum daily water temperature might be measured several times over the course of a summer to assess whether summer temperatures might be an important limitation to the quality of fish habitat under the existing conditions. However, if water temperatures are measured over several years to determine the effect of upstream management activities or climatic variations, this is clearly monitoring. The overlap in the definitions of assessment, inventory, and monitoring means that in some cases the primary distinguishing feature of monitoring will be the intent to assess change rather than the number or type of measurements.

Often an assessment or inventory serves as the first step towards establishing a monitoring project. Knowledge of the spatial and temporal variability is essential to developing an efficient monitoring plan (Chapter 3). To the extent that inventory and assessment techniques overlap with monitoring procedures, these *Guidelines* can help with the conceptual problems of deciding what, where, and how to inventory or assess water quality.

A number of federal and state agencies have defined the different types of monitoring carried out by their particular organization (e.g., Potyondy, 1980; Solomon, 1989). Unfortunately these definitions are not consistent, and this has often resulted in semantic confusion. In most cases a clear statement of the purpose of the monitoring will be the best method of defining the type of monitoring, and it then is simply a matter of attaching a mutually agreeable label to that particular type of monitoring. For the purposes of this document, the following types of monitoring are defined:

1. *Trend monitoring.* In view of the definition of monitoring, this term is redundant. Use of the adjective "trend" implies that measurements will be made at regular, well-spaced time intervals in order to determine the long-term trend in a particular parameter. Typically the observations are not taken specifically to evaluate management practices (as in type 4), management activities (as in type 5), water quality models (as in type 6), or water quality standards (as in type 7), although trend data may be utilized for one or all of these other purposes.
2. *Baseline monitoring.* Baseline monitoring is used to characterize existing water quality conditions, and to establish a data base for planning or future comparisons. The intent of baseline monitoring is to capture much of the temporal variability of the constituent(s) of interest, but there is no explicit end point at which continued baseline monitoring becomes trend monitoring. Those who prefer the terms "inventory monitoring" and "assessment monitoring" often define

them such that they are essentially synonymous with baseline monitoring. Others use baseline monitoring to refer to long-term trend monitoring on major streams (e.g., Potyondy, 1980).

3. *Implementation monitoring.* This type of monitoring assesses whether activities were carried out as planned. The most common use of implementation monitoring is to determine whether Best Management Practices (BMPs) were implemented as specified in an environmental assessment, environmental impact statement, other planning document, or contract. Typically this is carried out as an administrative review and does not involve any water quality measurements. Implementation monitoring is one of the few terms which has a relatively widespread and consistent definition. Many believe that implementation monitoring is the most cost-effective means to reduce nonpoint source pollution because it provides immediate feedback to the managers on whether the BMP process is being carried out as intended (Section 1.4). On its own, however, implementation monitoring cannot directly link management activities to water quality, as no water quality measurements are being made.
4. *Effectiveness monitoring.* While implementation monitoring is used to assess whether a particular activity was carried out as planned, effectiveness monitoring is used to evaluate whether the specified activities had the desired effect (Solomon, 1989). Confusion arises over whether effectiveness monitoring should be limited to evaluating individual BMPs, or whether it also can be used to evaluate the total effect of an entire set of practices. The problem with this broader definition is that the distinction between effectiveness monitoring and other terms, such as project or compliance monitoring, becomes blurred.

To minimize confusion within this document, effectiveness monitoring will be used in the narrow sense of evaluating individual management practices, particularly BMPs (Section 1.4). Monitoring the effectiveness of individual BMPs, such as the spacing of water bars on skid trails, is an important part of the overall process of controlling nonpoint source pollution (Sections 1.4 and Chapter 2). However, in most cases the monitoring of individual BMPs is quite different from monitoring to determine whether the cumulative effect of all the BMPs results in adequate water quality protection. Evaluating individual BMPs may require detailed and specialized measurements best made at the site of, or immediately adjacent to, the management practice. Thus effectiveness monitoring often occurs outside of the stream channel and riparian area, even though the objective of a particular practice is intended to protect the designated uses of a water body. In contrast, monitoring the overall effectiveness of BMPs usually is done in the stream channel,

and it may be difficult to relate these measurements to the effectiveness of individual BMPs.

5. *Project monitoring.* This type of monitoring assesses the impact of a particular activity or project, such as a timber sale or construction of a ski run on water quality. Often this assessment is done by comparing data taken upstream and downstream of the particular project, although in some cases, such as a fish habitat improvement project, the comparison may be on a before and after basis. Because such comparisons may, in part, indicate the overall effectiveness of the BMPs and other mitigation measures associated with the project, some agencies consider project monitoring to be a subset of effectiveness monitoring. Again the problem is that water quality is a function of more than the effectiveness of the BMPs associated with the project.
6. *Validation monitoring.* Since the issue of validating water quality standards is beyond the scope of this document, validation monitoring in these *Guidelines* is discussed primarily with regard to the quantitative evaluation of a proposed water quality model to predict a particular water quality parameter. In keeping with the basic principles of modeling (e.g., James and Burges, 1982), the data set used for validation should be different from the data set used to construct and calibrate the model. This separation helps ensure that the validation data will provide an unbiased evaluation of the overall performance of the model. The intensity and type of sampling for validation monitoring should be consistent with the output of the model being validated.
7. *Compliance monitoring.* This is the monitoring used to determine whether specified water-quality criteria are being met. The criteria can be numerical or descriptive. Usually the regulations associated with individual criterion specify the location, frequency, and method of measurement.

It should be emphasized that these seven types of monitoring are not mutually exclusive. Often the distinction between them is determined more by the purpose of monitoring than by the type and intensity of measurements. Regular sampling of coliform bacteria to meet health standards, for example, will produce data that also can be used to indicate long-term trends. Table 1 is a broad classification of monitoring types according to the parameters being measured, the frequency of monitoring, the duration of monitoring, and the intensity of data analysis. At this point no consensus exists on the definitions of monitoring types, and this, together with the proliferation of monitoring terminology, means that each monitoring plan should explicitly define the monitoring terminology being used.

These *Guidelines* are not equally applicable to all seven monitoring types as defined above. Most of the parameters used for trend, baseline, and project monitoring are explic-

Part I

Table 1. General characteristics of monitoring types.

Type of monitoring	Number and type of water quality parameters	Frequency of measurements	Duration of monitoring	Intensity of data analysis
Trend	Usually water column	Low	Long	Low to moderate
Baseline	Variable	Low	Short to medium	Low to moderate
Implementation	None	Variable	Duration of project	Low
Effectiveness	Near activity	Medium to high	Usually short to medium	Medium
Project	Variable	Medium to high	>Project duration	Medium
Validation	Few	High	Usually medium to long	High
Compliance	Few	Variable	Dependent on project	Moderate to high

itly considered, and the general discussion on developing a monitoring plan is directly relevant. On the other hand, implementation monitoring generally does not involve water quality measurements, and so the *Guidelines* are less applicable. Effectiveness monitoring of individual BMPs also may use different parameters than the ones discussed in these *Guidelines*.

Since validation monitoring is used to evaluate model accuracy, the parameters to be measured are defined by the model output. Usually these will correspond to some of the monitoring parameters reviewed in this document, but this may not necessarily be the case. Similarly, the parameters and procedures for compliance monitoring usually are specified by the regulating agency. Some standards are written in qualitative language, and in such cases several different procedures might be used to assess a broadly defined standard such as "biological integrity." Nevertheless, most of the constituents incorporated in compliance monitoring projects in forested areas are included in these *Guidelines*.

Most water quality monitoring projects will involve more than one of the types of monitoring defined above. The integration of several monitoring types into one project usually is due to multiple objectives. As suggested previously, distinct objectives attained through different types of monitoring do not necessarily require distinct and independent data collection efforts. If the monitoring objectives are clearly specified, one usually finds considerable overlap in terms of the data needs, and recognition of this can result in considerable cost savings.

Box 2—the first of five case studies presented in the *Guidelines*—is an overview of ongoing water quality monitoring efforts in the Bull Run watershed near Portland, Oregon. This particular project has been subjected to considerable scrutiny by several parties with diverse interests, and it recently underwent a thorough technical review (Aumen et al., 1989). Although it can be argued that the monitoring efforts on the Bull Run watershed are relatively unique in terms of their cost and intensity, the ongoing revisions in the monitoring project have much broader implications. Of particular interest is the reallocation of

monitoring effort from effectiveness monitoring to implementation monitoring.

1.4. LEGAL BACKGROUND

The different types of monitoring have evolved partly in response to the changing objectives and legal requirements for water quality monitoring. Passage of the Federal Water Quality Act of 1965 led to the widespread adoption of instream water quality standards. This stimulated state and local agencies to initiate more intensive monitoring programs, but these were oriented more towards meeting the legal requirements than facilitating management decisions (Sanders and Ward, 1979).

In 1972 the Federal Water Pollution Control Act established a regulatory system for point sources of water pollution. This added a permit system and self-monitoring of effluent discharge to existing instream monitoring efforts. A national goal that all waters should be fishable and swimmable was established.

Section 208 of the 1972 law recognized that nonpoint sources could adversely affect water quality and should be controlled. States were required to prepare plans for controlling nonpoint sources, although implementation was voluntary (Hohenstein, 1987). The primary mechanism for regulating nonpoint sources is by adopting and implementing BMPs (Best Management Practices). In general terms BMPs are defined as those practices, or combination of practices, that are practical and effective in preventing or reducing pollution from nonpoint sources to levels compatible with water quality goals (Lynch and Corbett, 1990). The current EPA definition of BMPs is as follows:

Methods, measures or practices selected by an agency to meet its nonpoint control needs. BMPs include but are not limited to structural and non-structural controls and operation and maintenance procedures. BMPs can be applied before, during and after pollution-producing activities to reduce or

Box 2. CASE STUDY: BULL RUN WATERSHED, OREGON¹

The 275-km² Bull Run watershed lies within the Mt. Hood National Forest about 50 km east of Portland on the west side of the Cascades. By law the principal management objective is to provide "pure, clear, raw, and potable" water for the Portland metropolitan area. The water is of exceptionally high quality and is chlorinated but not filtered before it enters into the municipal supply system. This means that water quality is of utmost concern, and the Bull Run watershed has the most intensive water quality monitoring program of any municipal watershed in the United States. In order for management to protect water quality, public access is restricted and activities such as timber harvest are carefully controlled.

The three monitoring objectives are as follows: (1) to assess the effects of management activities on water quality, (2) to monitor compliance with raw water quality standards, and (3) to investigate physical processes in order to improve the predictability of watershed response to management activities and climatic events. The types of monitoring conducted to meet these objectives are implementation, effectiveness, validation, and trend monitoring.

Implementation monitoring is the process of ensuring that the site-specific management requirements were carried out as planned. Effectiveness monitoring evaluates whether the management practices, including BMPs, adequately protected water quality. As defined by the Bull Run project, effectiveness monitoring consists of inventory monitoring and water quality sampling. Inventory monitoring measures those instream, riparian, and upslope characteristics that can affect water quality. Examples include large woody debris, stream shading due to the riparian canopy, and revegetation rates on exposed ground.

Water quality and discharge are measured at five key stations. Four of these are located at the mouths of the major tributaries to the water supply reservoir, and the fifth is located at the intake to the diversion facility. Another 14 source-search stations are used to monitor the larger streams and the main reservoir. Samples from these stations help determine the source of any high value recorded at a key station. Short-term monitoring stations are set up as needed to monitor the water quality effects of management activities such as salvage logging, prescribed burning, and road construction and maintenance. These project monitoring stations are preferably located upstream and downstream of the management activity, although sometimes a paired-watershed design is utilized.

Water quality standards have been established at the key stations for 39 variables. Discharge, temperature, and conductivity are measured continuously, while pH, color, turbidity, conductivity, suspended sediment, and several bacteriological indicators are measured at least weekly. The water quality standards are based on historical data, and a series of parametric and non-parametric statistical tests are used to determine when a particular value exceeds the standards. Any deviation must be evaluated and explained through a systematic procedure involving the source-search stations, additional investigations, and administrative procedures. Most deviations stem from the fact that the relatively short water quality record does not adequately reflect the more extreme climatic events.

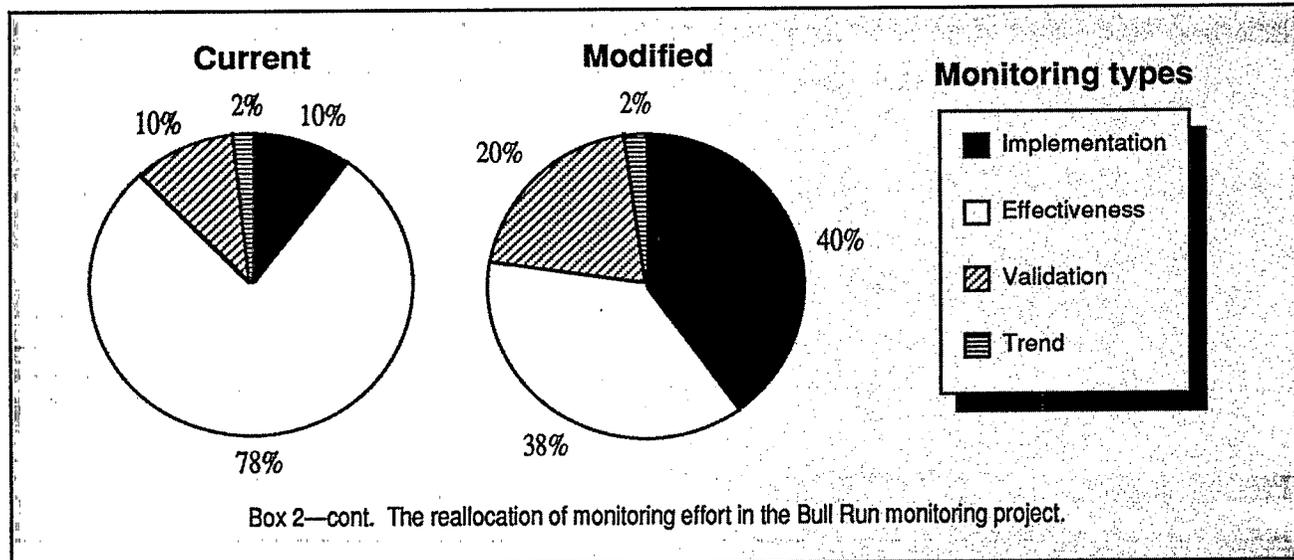
A recent technical review provides an excellent case study of the design and operation of this intensive water quality monitoring program (Aumen et al., 1989). Specific recommendations that may be more widely applicable include the following:

- increased emphasis on on-site monitoring during the course of the management activity, as this is the most effective method for minimizing adverse impacts of management activities on water quality;
- elimination of the source-search stations because they had not proven useful in detecting the cause of deviations, and they absorbed a large proportion of the monitoring resources;
- fewer, but more intensive, project monitoring sites;
- increased reservoir monitoring to detect subtle long-term effects;
- a shift away from sampling at equal time intervals to flow-based sampling with special emphasis on storm events;
- more emphasis on data interpretation and understanding watershed processes and functions; and
- altering procedures to allow a more flexible response to often meaningless deviations from the raw water quality standards.

The attached graph indicates how these recommendations have resulted in a reallocation of monitoring effort among the different types of monitoring being used on the Bull Run watershed. Previously about 75% of the effort was directed towards effectiveness monitoring, whereas the revised project places much more emphasis on implementation and validation monitoring.

¹Sources: Aumen et al., 1989; McCammon, 1989.

(Continued on p. 10)



eliminate the introduction of pollutants into receiving waters (CFR, 1990).

Some specific examples of BMPs include the appropriate placement of waterbars on skid trails, stabilization and treatment of cut and fill slopes, and seasonal restrictions on road use and log skidding. Certain BMPs may be certified as approved practices for controlling nonpoint source pollution by state water quality control agencies. In the case of forestry, Alaska, Idaho, Oregon, and Washington all have Forest Practice Acts which include specific BMPs in the associated rules and regulations. Each state water quality agency then certifies these BMPs as appropriate nonpoint source controls under the authority of the Clean Water Act. If water quality objectives are not met despite the proper implementation of BMPs, states can de-certify BMPs and require more stringent measures to minimize adverse effects on water quality.

The state water quality agencies also may certify BMPs for those federal agencies which are conducting land-disturbing activities. In general, these agency-derived BMPs must meet or exceed the relevant state requirements specified under legislation such as a state Forest Practices Act. Often state certification of BMPs submitted by a federal agency leads to a delegation of responsibility to that federal agency to protect and restore those water bodies under its jurisdiction. Such responsibility is subject to approval and review by the state water quality agency, and this in turn is dependent upon the ongoing monitoring program conducted by the federal agency.

A second mechanism to control nonpoint source pollution is through water quality standards. Water quality standards are legal requirements combining the designated uses of water with the numerical criteria necessary to protect those uses (see Section 4.2 for further discussion of the designated uses of water). Water quality criteria are either numeric

limits or narrative descriptions of water quality. The objective of establishing specific criteria is to protect human health and aquatic life, as well as the designated uses of water (EPA, 1988). EPA has recommended specific criteria for nearly one hundred water quality parameters, including bacteria, color, nutrients, metals, and a wide range of chemicals (EPA, 1986). Most of these criteria are expressed in numeric terms and have been adopted by the states in order to protect human health and aquatic life.

In the case of nonpoint pollution due to forestry activities, the use of water quality standards is hindered by the limited number of criteria which are applicable. Of the nearly one hundred criteria set out in EPA's Quality Criteria for Water (EPA, 1986), forest management activities are likely to affect only a few—dissolved oxygen, temperature, turbidity, suspended solids, and perhaps nitrate-nitrogen. Yet forestry activities are known to affect numerous other parameters which do not have EPA-specified criteria, but which have great significance for aquatic and riparian ecosystems (Salo and Cundy, 1987; Raedeke, 1988). Examples of these other parameters include the amount and type of large woody debris in stream channels, the quality of the streambed material for spawning, the amount of pool habitat, and the type and density of riparian vegetation (Part II, Chapter 5 and Section 6.2).

The primary role of water quality standards in regulating nonpoint source pollution has been affirmed by court decisions stating that water quality standards must be used to evaluate whether BMPs are effective (Hohenstein, 1987). Again monitoring is the basis for determining if existing water quality meets the relevant standard(s), and whether BMPs are effective.

The third mechanism for regulating water quality in forested areas is the antidegradation policy. This policy has three tiers or levels of protection (Fig. 1), and some form of monitoring is required for the successful implementation of

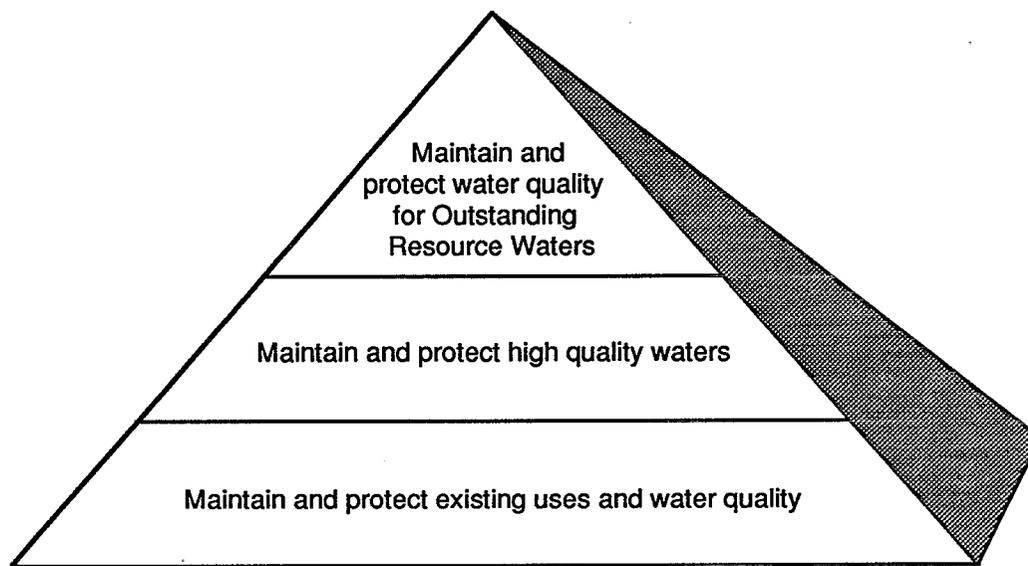


Figure 1. The three levels of the antidegradation policy for protecting existing uses and maintaining or improving existing water quality (EPA, 1988).

each tier. Basically EPA requires the states to place water bodies into one of these three tiers, with each tier having a different level of protection against the degradation of water quality. The lowest tier or level of protection requires that existing instream uses be fully supported (i.e., water quality is not limiting the existing uses). Monitoring for this purpose could vary from a periodic qualitative assessment of recreational suitability to quantitative measurements of parameters such as salmonid egg survival or intergravel dissolved oxygen.

The second tier or middle level of protection applies to those waters which have a water quality level higher than that necessary to support recreation and the propagation of fish and other wildlife. In these water bodies a lowering of water quality is allowed only if: (1) the state determines that this is necessary to accommodate important economic or social development, and (2) the decision is based on a review and comment process by the public and state and local agencies. However, the allowed degradation cannot adversely affect the existing uses as specified by the first tier of the antidegradation policy. Monitoring for this purpose may need to be more quantitative and statistically based as discussed in Chapter 3.

The third and highest tier of the antidegradation policy applies to those waters designated by the states as Outstanding Resource Waters. Such designation means that no degradation in water quality is allowed. Typically Outstanding Resource Waters are the highest quality waters in the state, and they have some special characteristics to justify this high level of protection. Water bodies in state or national parks, or water bodies which are part of the National Wild and Scenic River System, most often are considered for designation. Again monitoring at this third tier may require

quantitative measurements and an appropriate statistical design to detect a specified level of change.

Each of these three main mechanisms for controlling nonpoint source pollution—BMPs, water quality standards, and antidegradation—were further enhanced by the 1987 amendments to the Clean Water Act. In particular, Section 319 required the states to identify those water bodies which do not meet water quality standards due to nonpoint pollution. Each state then must develop a program to improve water quality to the point that water quality standards are met. These programs must be approved by EPA, although the actual implementation is carried out by the state agency responsible for water quality.

For those water bodies that still do not meet water quality standards, despite the implementation of point source controls and effective BMPs, a load allocation process may be initiated (bottom of Fig. 3, p. 15). Currently each state establishes its own priority list for initiating the load allocation process according to the value of the designated uses and the risk of damage to those uses.

This load allocation process is discussed in Section 2.1, but basically it is a relatively data-intensive procedure which may require detailed monitoring. The first step is to identify the constituent(s) of concern, and determine the frequency and timing of water quality violations. The loading capacity of the water body for each constituent that violates water quality standards must be quantitatively assessed, and then a safety factor is subtracted from this loading capacity. Further subtraction of the contribution of natural sources yields the Total Maximum Daily Load (TMDL). This is the amount of pollution which can be contributed by anthropogenic activities, and it is allocated among all the point sources ("wasteload allocation") and nonpoint sources ("load

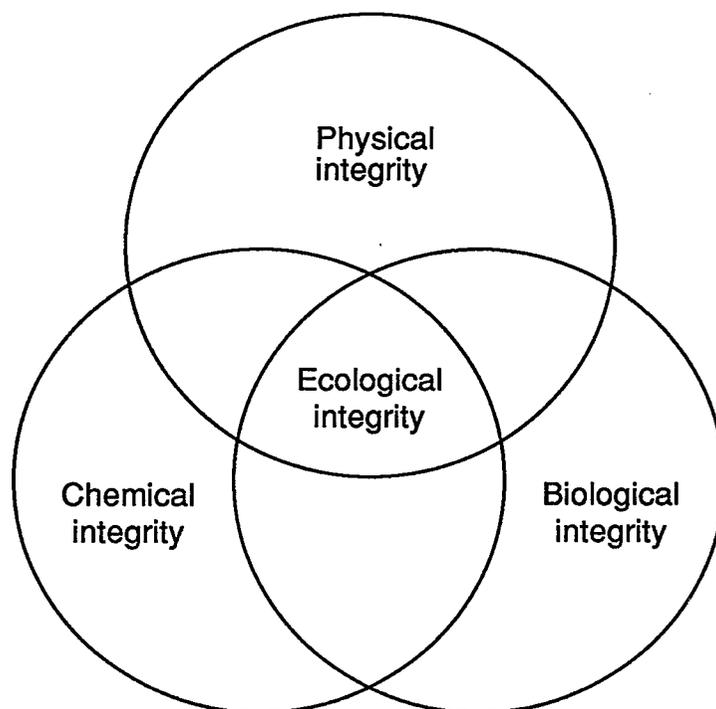


Figure 2. Ecological integrity is attainable when chemical, physical, and biological integrity occur simultaneously (EPA, 1990).

allocation"). In equation form,

$$\text{Loading capacity} - \text{safety factor} - \text{contribution from natural sources} = \text{TMDL}$$

$$\text{TMDL} = \text{wasteload allocation} + \text{load allocation}$$

To date the load allocation process has been developed for only a few water bodies in the Pacific Northwest, although it undoubtedly will be more widely applied in the future.

In summary, a variety of mechanisms have been developed to control nonpoint source pollution, and there will be continuing adjustments and additions to these regulatory tools in the future. To a certain extent it is this variety of objectives and approaches which has led to the proliferation of monitoring types and the confusion over monitoring terminology (Section 1.3). For example, compliance monitoring usually refers to the monitoring associated with meeting numerical water quality criteria and the limits specified in point source discharge permits. Implementation and effectiveness monitoring often are associated with the process of implementing and evaluating BMPs, but effectiveness monitoring also could apply to the evaluation of specific pollution control programs. Trend monitoring is necessary for the successful application of the antidegradation policy.

In late 1985 EPA recognized the changing needs in water quality monitoring and initiated a study of its surface water quality monitoring efforts. Several specific needs were identified (EPA, 1987), and these still are defining some of the current directions in water quality monitoring.

The first need was to develop and use biological monitoring techniques as well as the traditional physical and chemical water quality parameters. As shown conceptually in Figure 2, the biological integrity is one component of the ecological integrity, and biological monitoring is needed to evaluate the biological integrity. Hence the emphasis on developing biological criteria stems from the need for improved techniques to evaluate the condition of water bodies, as well as the need to more directly relate water quality criteria to designated uses (EPA, 1990). EPA is now working with the states to develop narrative biological criteria (EPA, 1990), and numerical biological criteria are likely to be developed subsequently.

The Rapid Bioassessment Protocols for aquatic macroinvertebrates and fish (Part II, Sections 6.2 and 6.3, respectively) are expected to serve as the prototype techniques for assessing and defining the biological integrity of streams. However, the establishment of biological criteria for streams in the Pacific Northwest and Alaska is likely to be an extended A second recommendation of the EPA monitoring study was that water quality monitoring programs should aim to demonstrate the results of water pollution control efforts

(EPA, 1987). Such data are needed to create a feedback loop for guiding future management decisions, and to justify the resources being spent on water pollution control efforts. This need to link water pollution control efforts and water body condition has important implications for the selection of monitoring parameters and the type of monitoring projects being undertaken. In particular, this objective suggests a move away from the traditional fixed-site monitoring stations in downstream locations, as water quality changes at these downstream sites tend to be smaller in magnitude and more difficult to relate to management actions. Locations higher in the watershed, where the monitoring data can be

more easily linked to specific management actions, are more likely to be emphasized in the future. The need to document change may also stimulate a shift away from the traditional physical and chemical parameters to parameters that are more sensitive to management activities and which can be directly related to the designated uses. An objective of these *Guidelines* is to facilitate this change in emphasis.

A third challenge identified in the EPA review was to identify and characterize pollutants from nonpoint sources (EPA, 1987). The selection and review of monitoring parameters in these *Guidelines* can be considered as one component of EPA's response to this challenge.

2. CONTEXT AND STRUCTURE OF MONITORING PROJECTS

The previous Chapter defined seven types of water quality monitoring and discussed the development of nonpoint water pollution control programs in the U.S. The purpose of this Chapter is to show how the different types of water quality monitoring projects fit into the general legal framework described in Section 1.4. This functional description of water quality monitoring will then be used as the basis for outlining the overall structure of water quality monitoring projects.

Chapters 3, 4, and 5 expand upon the broad outlines presented in this chapter. Chapter 3 reviews the principles of statistical design and sampling as they apply to water quality monitoring. In Chapter 4 more specific guidance is provided on developing water quality monitoring projects. This includes detailed discussions of driving factors such as the specification of monitoring objectives, the designated uses of the water body to be monitored, the type of management activities being carried out, and the physical environment. Chapter 5 integrates the information from Parts I and II of the *Guidelines* in order to suggest parameters for monitoring the effects of forestry activities on streams in the Pacific Northwest and Alaska.

2.1 LEGAL CONTEXT OF WATER QUALITY MONITORING EFFORTS

The most important aspects of the laws regulating nonpoint sources of water pollution (Section 1.4) can be summarized as follows. First, there is a broad mandate to ensure that the designated uses of water are protected, and to make all the nation's waters fishable and swimmable. Second, Section 208 of the 1972 Water Pollution Control Act required every state to establish effective Best Management Practices (BMPs) to control nonpoint source pollution. The establishment and revision of BMPs for various management activities is an ongoing, iterative process.

Third, Section 303(d) of the 1972 Water Quality Act required the states to list those water bodies that have designated uses impaired by water quality. The states now have begun a process of establishing relative priorities among these water bodies, developing plans for controlling the specific types of pollution limiting the designated uses, and implementing these plans as funds permit. To date the primary emphasis has been on controlling point sources through the wasteload allocation process, but for many water bodies there is a need to incorporate load allocations for nonpoint sources.

Fourth, the states recently have been encouraged to develop plans to protect or restore those water bodies that are impaired or threatened by nonpoint sources of pollution. Funds to implement these plans have become available from EPA under the authority of Section 319 of the Water Quality Act of 1987. The fifth and final mechanism to address nonpoint sources of pollution is the antidegradation policy, which is designed to protect water quality when the water quality already is higher than existing standards. It also is being implemented on a state-by-state basis.

To a large extent the success of each of these policies and pollution control mechanisms is dependent on monitoring. Water quality monitoring data and information on the designated uses are needed to determine which policy is to be applied, the relative priorities for action, and the development of specific plans for remediation. High quality monitoring data are increasingly needed to evaluate the effectiveness of pollution control efforts and thereby justify the expenditure of public funds. There also will be an increasing need for water quality monitoring data to compare the different procedures for controlling nonpoint source pollution. All these different objectives and data needs have helped spur the observed proliferation in monitoring types and terminology (Section 1.3). A schematic overview of how water quality standards, BMPs, and the load allocation process are used to control nonpoint sources of pollution is presented in Figure 3. Figure 3 also shows how the major

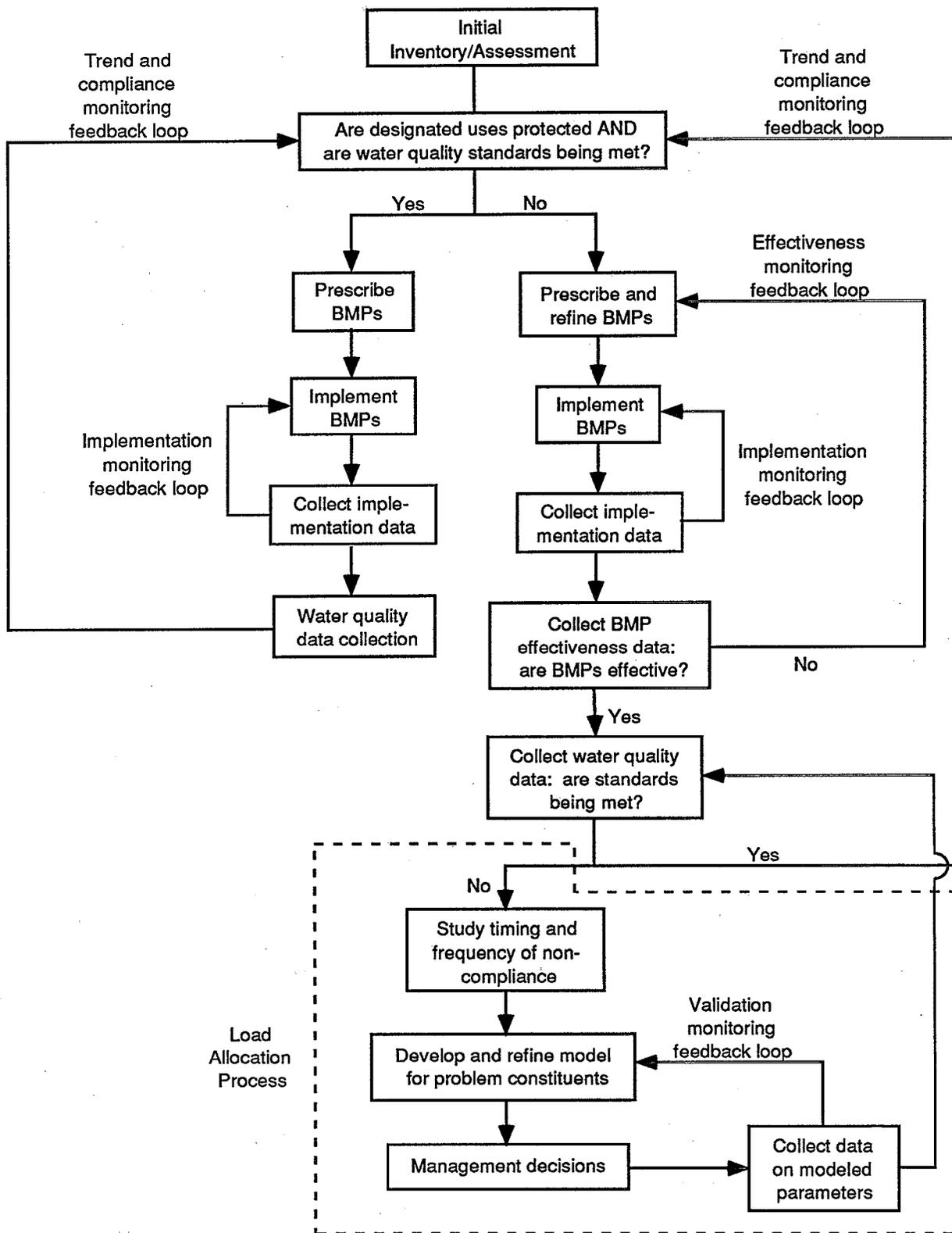


Figure 3. Flow diagram for monitoring and controlling nonpoint sources of pollution.

Part I

monitoring types defined in Section 1.3 fit into these three components of the current regulatory structure.

As indicated in Figure 3, the first step in developing a plan to control nonpoint source pollution is to determine whether water quality is (1) limiting the designated uses for that water body, and (2) meeting water quality standards. Such an evaluation necessitates an initial set of water quality measurements (baseline monitoring). If the designated uses are not impaired and the standards are being met, the procedures shown on the left-hand side of Figure 3 are applied. Basically this involves the routine application of BMPs and regular water quality measurements, with monitoring being an essential component of both of these activities.

The effective application of BMPs requires regular implementation monitoring (i.e., determining that the BMPs were applied as planned). This information must be fed back to managers in order for them to assess whether the BMP planning and implementation process is working. This implementation monitoring feedback loop (Fig. 3) is a crucial link in helping to ensure that BMPs are properly integrated into ongoing management activities, and generally is regarded as one of the most cost-effective means for controlling nonpoint source pollution.

Similarly, continued water quality monitoring is required to ensure that (1) the existing and designated uses of water continue to be unimpaired, (2) the applicable water quality standards continue to be met, and (3) there is no degradation of water quality. In theory these three goals all fall under the umbrella of meeting water quality standards, but in practice these goals often must be considered separately. According to the definitions in Section 1.3, a combination of trend and compliance monitoring is needed to achieve these goals. The trend and compliance monitoring feedback loop on the left-hand side of Figure 3 emphasizes that these data must be evaluated on a continuing basis, and a degradation in water quality probably will force a change in the procedures being used to limit nonpoint source pollution.

If the initial assessment of water quality indicates that the designated uses are impaired, or that the standards are not being met, the process on the right-hand side of Figure 3 is followed. Again the first management action is to prescribe and refine BMPs, as this is the primary means to protect water quality from nonpoint source pollution (Section 1.4). The implementation of BMPs must be regularly monitored to ensure that the observed water quality problems are not just a result of substandard field operations.

Note that this implementation monitoring is not necessarily limited to internal reviews. Many states have conducted extensive project reviews to assess the actual application of forest practice rules and BMPs. In most of these reviews, an interdisciplinary review team has conducted on-site evaluations of selected projects. Typically no measurements of water quality are made, although the field review team may qualitatively evaluate management impacts on stream channels. On the basis of these field reviews, the

interdisciplinary team makes recommendations regarding specific management practices and suggests procedures for ensuring that these are fully implemented. Such qualitative field reviews often are considered to be the most cost-effective means for protecting water quality, as current BMPs are believed to adequately protect water bodies from rapid and obvious degradation, and these reviews directly address the problem of implementation (e.g., Idaho Dept. of Health and Welfare, 1989; NCASI, 1988).

The second type of monitoring shown on the right-hand side of Figure 3, continuing trend and compliance monitoring, is similar to the trend and compliance monitoring discussed for unimpaired water bodies. Such monitoring provides the data needed to determine if (1) water quality standards are continuing to be violated, (2) the designated uses are impaired, and (3) water quality is improving. Again these data must be regularly analyzed and evaluated to determine what additional control measures should be undertaken. The water quality data also can help indicate the effectiveness of BMPs in protecting water quality, and for this reason some agencies regard the trend and compliance monitoring shown in Figure 3 as another type of effectiveness monitoring.

Continuing water quality problems often trigger a more intensive review of BMP effectiveness. As mentioned in Section 1.3, several approaches can be taken to BMP effectiveness monitoring. The simplest is a qualitative field inspection, which can be done individually (e.g., observing road drainage problems during storm events) or as part of a formal review team. The review team process is similar to that already discussed for implementation monitoring, but for BMP effectiveness monitoring the review team also must attempt to qualitatively assess whether proper implementation of BMPs adequately protected the water bodies of concern.

As noted, water quality monitoring is a secondary, broad-scale method of evaluating BMP effectiveness. By definition this approach relies on inchannel measurements, although additional upslope observations are needed to determine the cause(s) of any observed change in water quality. The emphasis on inchannel measurements means that these *Guidelines* can be used to help formulate plans for monitoring the overall effectiveness of BMPs.

In some cases it may be difficult to determine the precise cause of a particular water quality problem. Since both water quality monitoring and the interdisciplinary review team approach tend to assess the overall effectiveness of BMPs, a third mode of BMP effectiveness monitoring—evaluating individual BMPs—has been used to obtain the necessary rigor. This mode usually involves detailed field measurements on replicated sites. For many BMPs, measurements must be made outside of the stream channel, as these will have the necessary sensitivity to the practice being evaluated, and be less subject to confounding factors than instream measurements. Although evaluating individual BMPs may be a more costly approach than monitor-

ing inchannel parameters, these more specific measurements generate the detailed information needed to modify or promulgate specific practices.

Typically several iterative cycles are needed to establish and refine BMPs, and this is the basis of the BMP monitoring feedback loop and the concept of adaptive management. In 1972 the Clean Water Act recognized that in some situations water quality standards would not be met, despite the regulation of point sources and the application of BMPs. In these cases the degradation of the designated uses is presumed to be due to the cumulative impact of the various sources. Control of the critical pollutant(s) then must be addressed through the load allocation process summarized in Section 1.4 and shown at the bottom of Figure 3.

The first step in the wasteload allocation process is to determine how often and when water quality criteria are being violated. These monitoring data are then combined with land use, point source, and watershed information to develop a water quality model for the constituents of concern. This model provides the technical basis for formulating the management decisions needed to reduce the pollutant load and bring the water body into compliance with the applicable water quality standards. The entire process is very data intensive, as it requires (1) identification of the pollutant sources; (2) determination of the times, frequency, and magnitudes of violation; (3) calibration and validation of the water quality model; and (4) continued monitoring to determine if management actions are effective. The difficulty in applying the wasteload allocation process (e.g., Ice, 1990) is one reason why this approach has been implemented for only a few water bodies in the Pacific Northwest and Alaska.

This review of the regulatory context of water quality monitoring illustrates the role of most of the major types of monitoring (baseline, implementation, effectiveness, trend, compliance, and validation) defined in Section 1.3 and the critical need for feedback loops. The presence of these different types of monitoring does not necessarily mean that there should be six distinct data sets. In many cases data from one type of monitoring can be utilized for other purposes. For example, the data needed for trend and compliance monitoring also can be used to evaluate the overall effectiveness of BMPs, or perhaps for validation monitoring. This overlap is why the monitoring types defined in Section 1.3 are distinguished more by their specific objectives than the particular monitoring technique or data collection methods. The fact that a single monitoring activity can serve several purposes also suggests that a carefully designed monitoring plan can substantially reduce the data collection costs.

Although effective feedback loops are critically important to the design and execution of monitoring projects, it is remarkable how often water quality data are collected but not analyzed, or are utilized in a relatively superficial manner. If the data are not being regularly analyzed, those monitor-

ing efforts by definition are only fulfilling an administrative requirement or political need. In such cases the resources being directed towards monitoring are being inefficiently utilized, and the data being collected have essentially no value because they are not being transformed into useful information for land managers, scientists, or the public regulatory agencies.

This suggests a failure in either defining the objectives (Section 4.1), or in designing and implementing the feedback loops. Feedback loops need to be explicitly incorporated into monitoring plans, and this needs to be done in several ways and at several different levels. For example, the resource specialist has to ensure from the start that time is allocated for analyzing and evaluating the data. Specialists and managers should recognize that data analysis actually can be a cost-effective process, as it can improve (1) the location and timing of data collection, (2) the choice of monitoring parameters, and (3) the appropriateness of the monitoring objectives. Rapid data analysis also can provide early feedback to adjust BMPs if additional protection is needed, or conversely identify situations where fewer controls will suffice to protect water quality.

On a different level, the establishment of an implementation monitoring project is an excellent means to bring together managers, operational personnel, and resource specialists. In all likelihood this team approach will facilitate the development of the other feedback loops as each member becomes more involved in the monitoring process and works for its success.

Although feedback loops are essential to developing an effective monitoring plan, in many cases it may be best to first conduct a pilot monitoring project. As discussed in Chapter 3, a pilot project provides much of the initial data needed to define a monitoring plan that is efficient in terms of its design and sampling procedure. A pilot monitoring project also allows time for personnel to become familiar with sampling devices and analytical equipment, thus improving the reliability of subsequent data. A pilot project also provides a set of test data for analysis and evaluation, which helps clarify the linkage between the water quality measurements and the monitoring objectives. In short, a test project forces one to go through each stage of developing and implementing a monitoring plan, but without a long-term commitment of resources. All too often a monitoring project, once established, takes on a life of its own and is difficult to modify even though it may not be meeting the original objectives. A pilot project is far easier to modify because it is conducted on a trial basis.

In almost every case the development of a water quality monitoring plan should be considered as an iterative process. It is unrealistic to expect that the monitoring parameters, sampling locations, sampling frequency, and measurement techniques will be optimal from the beginning. Each watershed and each monitoring project is different, and in the absence of *a priori* information on the statistical dis-

tribution of the parameters in time and space, no monitoring program will be optimal. Also, our knowledge of monitoring parameters and techniques will continue to change. Thus one should expect to refine the monitoring program over time as the data is collected and analyzed. On the other hand, a change in parameters or techniques could well preclude any statistical comparisons with earlier data—another reason why a pilot project should be conducted before a long-term monitoring project is initiated.

The case study of the South Fork of the Salmon River illustrates how a long-term monitoring project has adapted as needs changed and additional information became available. The common theme over the nearly 25 years of monitoring has been to focus on those parameters that provide specific, quantitative information on the limiting factors (spawning and rearing habitat) for the designated use of greatest concern (salmonid fisheries) at least cost (Box 3).

2.2 STRUCTURE OF A WATER QUALITY MONITORING PROJECT

Many of the key steps for defining and implementing a water quality monitoring project have been identified through the discussion in Section 2.1. Although the definition of the specific steps in developing a monitoring project tends to vary according to the author of the guidelines and the particular monitoring situation (e.g., Boynton, 1972), 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 4 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 4. 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 the 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 4 is one approach to optimize the process of developing a monitoring project.

The first step is to identify the general objectives, and this is 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, change 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 that data. Both sides also must be explicitly aware of the risks and uncertainties associated with monitoring in a highly variable environment (Section 3.2.3).

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 the specific objectives also requires some knowledge of the fluvial systems to be monitored (Section 4.8) and the likely impact of management activities (Section 5.1).

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 (Section 4.1).

Once the specific objectives have been formulated, the next step is to select the parameters to be measured (Chapter 5) and set out the protocols for collecting data and analyzing

Box 3. CASE STUDY: SOUTH FORK SALMON RIVER, IDAHO

The South Fork Salmon River is located in the mountains of Idaho in the west, central part of the state. The 1290-mile² watershed is characterized by steep slopes and shallow, coarse-textured, granitic soils that are extremely erodible. Almost all of the watershed is National Forest land. From 1945 to 1965, intensive logging and the associated road construction and maintenance accelerated erosion in the watershed. Extreme climatic events in 1964 and 1965 flushed much of the eroded material downstream. The deposition of the sand-sized material in the main channels resulted in extensive channel aggradation. Widespread concern about valuable salmon and steelhead fisheries led to a moratorium on logging activities, a broad-scale watershed rehabilitation program, and a long-term program to monitor spawning and rearing habitat conditions.

The initial monitoring program began in 1966 in the upper portions of the watershed, where most of the past logging activity had occurred. In each of the four major spawning areas, a series of 10-20 channel cross-sections were established to monitor long-term trends in the particle size distribution of the streambed surface. The particle size distribution of subsurface sediments was evaluated by taking samples at 20 randomly selected sites in each spawning area using a McNeil core sampler. Sets of five cross-sections spaced at 1-mile intervals were used to monitor changes in the particle size distribution of the streambed surface in rearing areas along the main channel. A series of 24 surveyed channel cross-sections evaluated changes in bed elevation over time: 6 of these were located in key spawning and rearing areas, with the remainder in representative pool and riffle areas. Additional monitoring data included: large-scale aerial photo surveys at about 5-year intervals, fixed photo points to document changes in streambed surface conditions, surveys of the depth of accumulated sand in key pools, snorkel surveys of juvenile fish populations in key rearing areas, and occasional water samples for chemical analysis to evaluate heavy metal outputs from mining activities in the headwaters of the basin.

By 1978 monitoring data showed that most of the accumulated sand deposits had been flushed out of the system. As a result, a new land management plan was developed that allowed for a cautious continuation of logging activities in the South Fork. However, all logging was contingent upon the continued improvement of fish habitat conditions as documented by an expanded set of monitoring activities. This monitoring was a three-phase effort to: (1) evaluate on-site effects of specific management activities; (2) assess changes in the tributary streams as a result of the renewed logging activities; and (3) monitor changes in aquatic habitat conditions in the main stem of the South Fork of the Salmon River. A technical monitoring committee was established to recommend procedures, evaluate results, and advise land managers about trends in habitat quality and river conditions. The committee membership included representatives from the two National Forests responsible for managing the watershed, the U.S. Forest Service Intermountain Research Station, the U.S. Fish and Wildlife Service, the Idaho Department of Fish and Game, and the Idaho Wildlife Federation.

On-site data collection was designed to assure that logging and road construction activities were performing as planned. Areas of concentrated erosion, such as road prism failures and drainage failures, were measured to document total soil loss. At selected sites in zero-order (headwater) basins, the amount of downslope soil movement was estimated by the use of detention basins.

Tributary channel monitoring has included measurements of embeddedness in disturbed and undisturbed basins. Some instantaneous discharge and sediment measurements have been made during high flow (spring snowmelt) periods. In the mainstream the previous monitoring efforts have been continued, with additional data being collected on embeddedness and the macroinvertebrate populations.

The technical monitoring committee has continued to evaluate the monitoring results and make recommendations for change as necessary. For example, additional tributary embeddedness data were collected when an initial survey suggested that the degree of embeddedness was directly proportional to the extent of road construction in the watershed. Changes in the mainstream monitoring efforts have included modifying the macroinvertebrate sampling locations and altering the location and sampling frequency of the surveyed channel cross-sections. Monitoring costs have been an important constraint, and these have been taken into account in the deliberations of the technical monitoring committee. In developing and evaluating the monitoring program, the committee also considered natural perturbations, including an oxbow breach in the South Fork of the Salmon, which caused extensive changes in channel morphology, wildfires, and localized floods within specific tributaries.

Sources: W. F. Megahan, Intermountain Forest and Range Experiment Station, U.S. Forest Service, Boise, Idaho; Torquemada and Platts (1988).

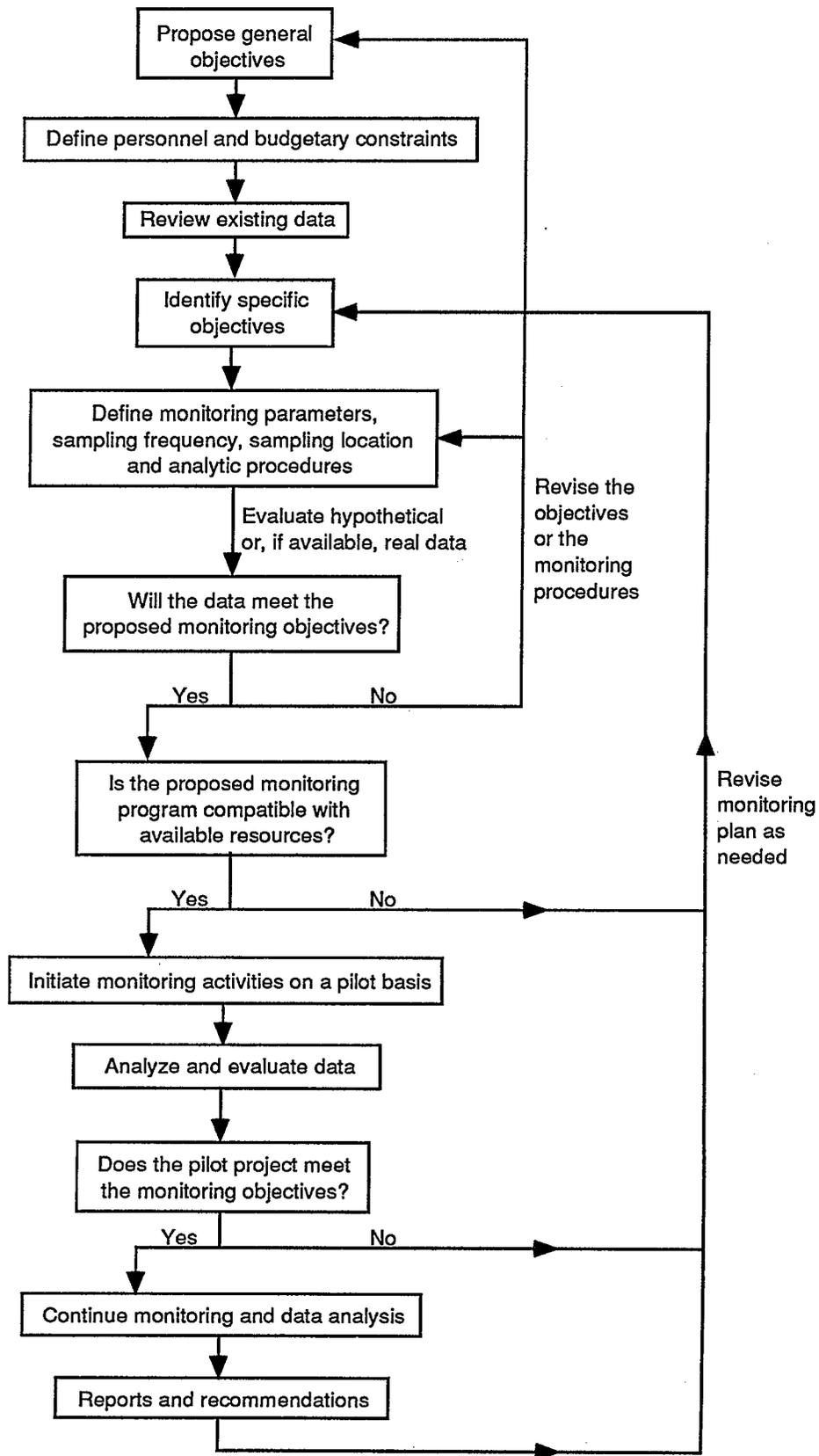


Figure 4. Development of a monitoring project.

field samples. Provision for outside analyses and repetitive samples is needed for quality assurance and quality control. The frequency, duration, and location of measurements (Chapter 3) will be determined by the objectives and the decisions with regard to the trade-offs among sample size, variability, risk, and uncertainty (Section 3.4.2).

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 4, 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 cost might be achieved by reducing the number of sampling sites, reducing the number of parameters to be monitored, or reducing the frequency of sampling (see Box 5, p. 37, for suggestions on how to reduce the cost of a monitoring project). 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 modification 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 it is 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 4, 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 4 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 the 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.

3. STATISTICAL CONSIDERATIONS IN WATER QUALITY MONITORING

3.1 RELEVANCE OF STATISTICS TO WATER QUALITY MONITORING

Statistics are an inherent component of nearly all water quality monitoring programs. In most cases a precise formulation of the monitoring objectives (Section 4.1) 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 either 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.

Other common monitoring objectives include the characterization of a parameter (baseline monitoring), determination of trends (trend monitoring), evaluation of models and standards (validation monitoring), and assessment with regard to a set standard (compliance monitoring). Each of these requires collecting and analyzing data. Statistics provides the scientific basis and procedures for studying numerical data and making inferences about a population based on a sample of that 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 programs 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 populations being sampled.

To take full advantage of these potential benefits, those responsible for preparing monitoring plans should consult with a statistician both 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 balance the relative risks and costs associated with different types of errors (Section 3.4.2). The managers 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 tests is not addressed in this document, as a number of texts provide a much more extensive review of experimental design and data analysis (e.g., Gilbert, 1987; Green, 1979; Sanders et al., 1987; and Zar, 1984). The books by Gilbert and Sanders are particularly noteworthy because they focus on the statistical methods for monitoring environmental pollution and the design of water quality monitoring networks, respectively. A series of papers by the U.S. Forest Service provides a particularly clear and simple explanation of the statistical aspects of water quality monitoring in forested areas (Ponce, 1980a,b), but these may not be as readily available. Two books that focus on nonparametric statistics are recommended: the classic, easily understood text by Siegel (1956) and the recent, more rigorous treatment by Daniel (1990).

3.2 STATISTICAL DESIGN IN WATER QUALITY MONITORING

3.2.1 GENERAL DESIGN AND REPLICATION

The overall design of a monitoring project is largely determined by the monitoring objectives (Section 4.1) and closely tied to the type of monitoring (Section 1.3). In many cases the design of the monitoring plan will determine the statistical procedures used to analyze the 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

sampling 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 designated 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 signifi-

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cant 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 Harned et al. (1981) for analyzing discharge-related parameters and Montgomery and Reckhow (1984) for analyzing serially correlated data.

A completely different approach is to model the time series sequence using the techniques developed by Box and Jenkins (1976). This requires a minimum of 50-100 data points collected at approximately equal time intervals, does not allow for missing data, and is more complex than the techniques mentioned previously (Gilbert, 1987; Montgomery and Johnson, 1976).

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 any 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) of 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 the 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 parameter being measured. An estimate of the variability is necessary to make any statistically based inference about the cause of an observed difference between the treated and control sites. Since in most cases sites are not replicated,

claims of cause and effect must be based on other information and not statistical testing (Hurlbert, 1984). Ideally data should be collected to document the processes responsible for the observed change.

Multiple pairs of treated and control sites, although costly, usually result in a high sensitivity to change. Both the control and treated sites are subject 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., Naiman 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 statistics 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 access 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 the variation among the different 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 inferences 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 usually 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 a 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 any 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 case study in Box 4 summarizes the objectives and corresponding statistical design for a water quality monitoring project in the Snohomish River basin (Washington). Although this particular project is focusing on the effects of commercial agriculture, the principles guiding the timing

and location of sampling are equally applicable to forestry and the other nonpoint sources of pollution discussed in these *Guidelines*.

The problem of separating site differences from treatment differences is particularly acute for many of the channel parameters reviewed in Part II, Sections 5.1-5.6. Channel cross-sections, pool parameters, and bed material particle size are all sensitive to environmental factors such as the local geology and landforms (Section 4.8), and they may exhibit considerable variation over relatively short distances. As discussed in Section 3.3., 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 availability 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 is 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

BOX 4. CASE STUDY: DESIGN OF A MONITORING PROJECT ON THE SNOHOMISH RIVER, WASHINGTON

In 1987 a 3-year water quality monitoring project was initiated on the Snohomish River basin in western Washington. The overall purpose of the project was to evaluate the effect of commercial agriculture on water quality in the major tributaries and the main stem of the Snohomish River. The three specific monitoring objectives were to (1) establish a baseline set of water quality data, (2) determine if there were any trends in water quality over the course of the study, and (3) determine the effect of commercial agriculture on water quality (Luchetti et al. 1987).

The statistical design is based on a comparison of water quality upstream and downstream from major agricultural areas. Fifteen paired sites were established, with 10 of these pairs on tributary reaches and 5 pairs located along the mainstem of the Snohomish River. The pairs varied considerably in terms of the size of the stream being monitored and the amount of agricultural activity between the upstream and downstream sampling locations. Measured water quality parameters included turbidity, dissolved oxygen, nitrates, orthophosphates, and fecal coliform.

Sampling was stratified by wet (November-March), dry (March-October), and storm periods. The wet and dry periods were sampled at 10 and 7 equal time intervals, respectively. Variations in discharge caused some of the wet and dry period samples to be placed in a different strata than originally intended. In each year three storms were sampled at 5 or 6 of the paired sites, and 3-5 separate water samples were taken from each site over the course of the storm.

The design of the monitoring plan allows for statistical testing of the differences between years, seasons, stream pairs, and location (upstream or downstream). When the monitoring is completed, data will have been collected on the type and extent of agriculture between each upstream and each downstream location, as well as the application of Best Management Practices (BMPs). By combining this land use information with the water quality data, the project expects to be able to qualitatively and quantitatively document the magnitude and impact of commercial agriculture on water quality.

Source: G. Luchetti, Surface Water Management Division, King County, Seattle, WA.

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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 (Section 3.4.2).

3.2.2 BENEFITS OF A 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 or 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 for 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, measurements might only be made on a random sample of 10% of the units. If this sample indicated a statistically significant difference, measuring the remaining units might be unnecessary, and a substantial savings 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 input 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 (Section 3.3). 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.

3.2.3 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 of 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 de-

pendence of the parameter 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 (Section 2.2). 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 (Part II, Section 4.5.). 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 the 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 control sites to changes at sites with management activities (Section 3.2.1). The unmanaged control sites provide an index of change due to some key factors, such as climate, and the removal of these factors increases the sensitivity of the monitoring. Problems with this approach include the additional costs 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 a 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 inchannel 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 BMPs. 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

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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 BMPs, 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 measures.

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.

3.3 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 numbers 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 days 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 variability 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 (Part II, Sections 6.3 and 6.4), while discharge is often used to establish strata for sampling sediment and the other physical and chemical constituents of water. Stratifi-

cation 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 on 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 costs 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. Trend monitoring usually is done at stable, accessible sites on major streams or rivers.

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 monitoring 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 spatially 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 classification systems at this time. Once the stream segments have been identified, an additional stratification into habitat units (Part II, Section 5.5) 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, geomorphically 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 (Part II, Section 4.6.2).

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 sample location and timing discussed above. Composite samples over time or space can represent a substantial savings 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.

3.4 PRINCIPLES OF STATISTICAL TESTING

3.4.1 ASSUMPTIONS AND DISTRIBUTIONS

Most of the better-known statistical tests have been developed for scalar 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

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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 the 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 positively 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 a frequency distribution. A plot of the raw data is 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 show 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 transformation 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 transformation 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 one standard deviation of the mean, 95% of the data falls within two standard deviations 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. Random sampling was discussed briefly in Section 3.3, and it is discussed in the statistics texts cited previously.
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 (Section 3.2). 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 control techniques (e.g., EPA, 1983) are an important means for reducing the pos-

sibility of non-random errors. Plots of the data and residuals can help identify unusual trends, and non-parametric tests can be used to assess whether the errors are significantly non-random.

3.4.2 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 Section 3.4.1, 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 one 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 increases 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 a 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 given the statistical objectives and the known variability of the parameter. Often information on the variability of a 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 (Chapter 2 and Section 3.2.2). 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 (see p. 33 and Fig. 6) (L. Conquest, Univ. Washington, pers. comm.).

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

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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 5. 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 level of confidence that the difference is real, and a lower probability that it was due to chance. The most common ways of obtaining a 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 5A 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 $\alpha = 0.05$, 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 5B 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 5C is similar to Figure 5B, except that the variability of populations A and B have 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 have 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 5D illustrates that a weaker level of significance will increase the 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 is monitoring for bacterial contamination in 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 than 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 level of significance.

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 β is commonly known as Type II error. Type II error can be

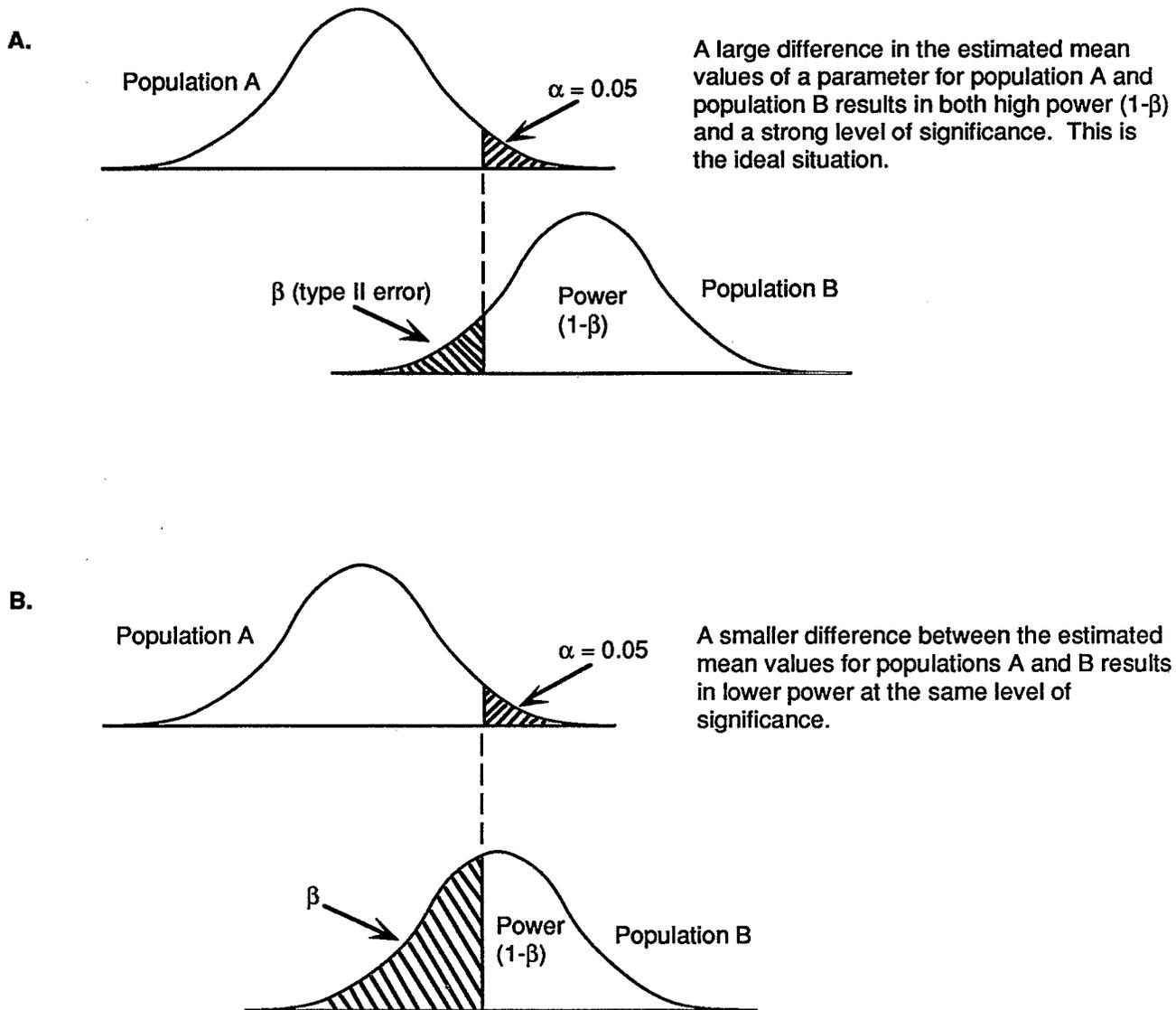


Figure 5. Schematic representation of the trade-offs among level of significance, power, and variability for two normally distributed populations. The figures assume a one-tailed t-test is being used to determine if a significant difference exists between the two populations.

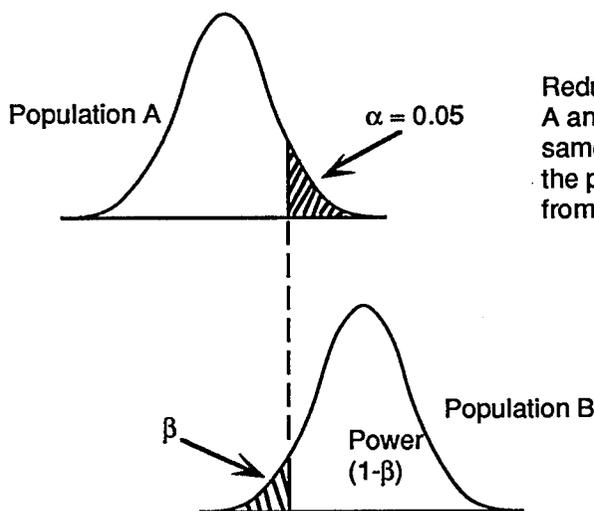
described as the probability of incorrectly concluding that two populations are the same when in fact they are different.

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

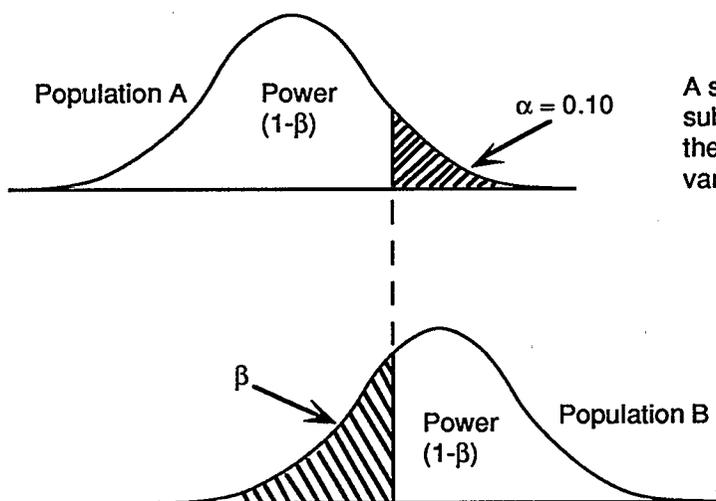
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C.



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 Fig. 5B.

D.



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

Figure 5—cont.

acceptable, this is 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 if 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

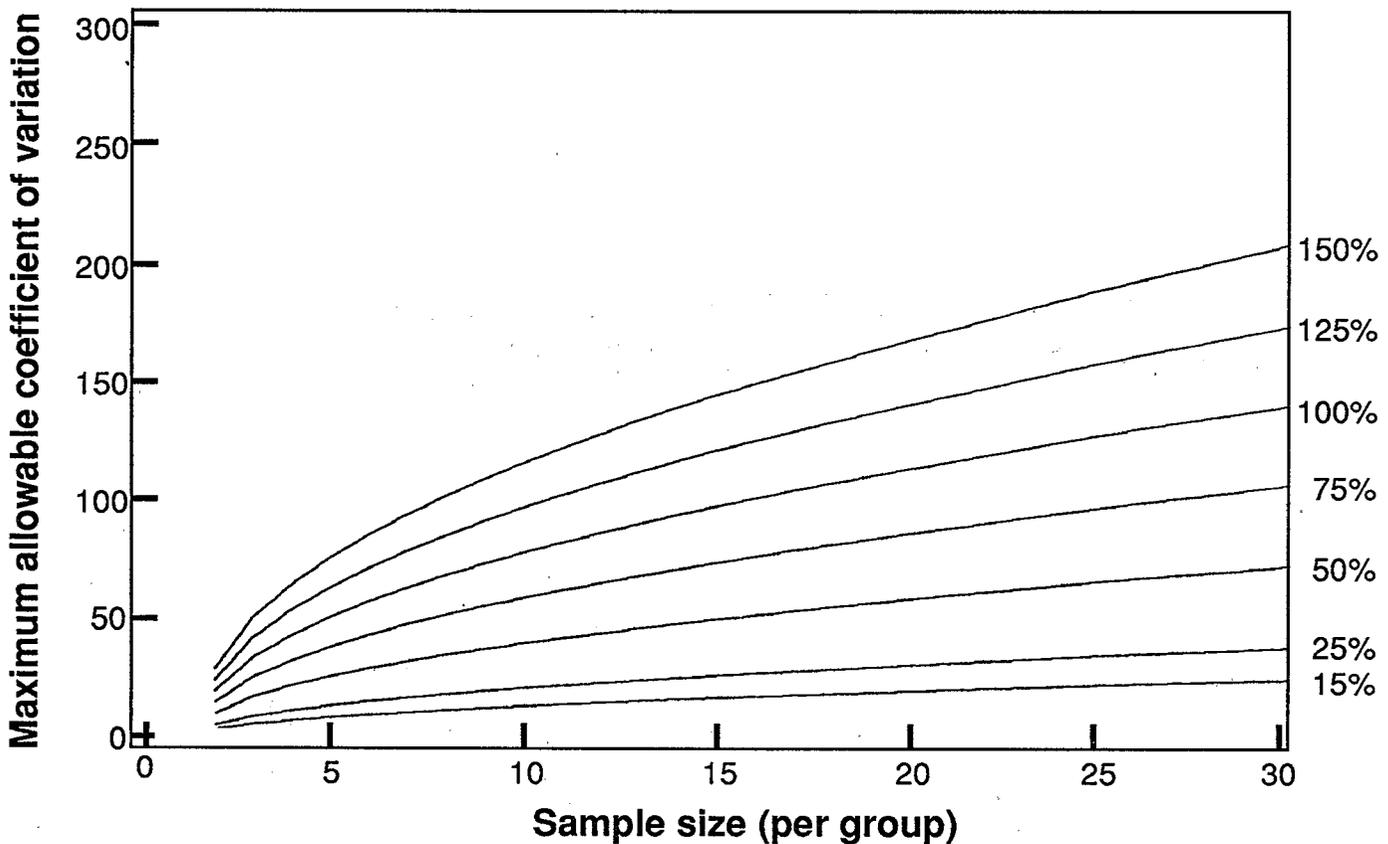


Figure 6. Maximum allowable coefficient of variation to detect changes ranging from 15 to 150%. Figure assumes a two-sample t-test is being used to detect change at a 5% level of significance and a power of 80%. The labeled curves show the minimum percent change that can be detected given a particular coefficient of variation for the parameter being measured and population sample size (figure courtesy of L. Conquest, Center for Quantitative Studies, University of Washington).

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 (Part II, Section 3.2).

Figure 6 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 6. In very rough terms, Figure

6 indicates that the treatment effect, expressed as 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 monitoring parameters (Chapters 4-5).

4. PRINCIPLES OF DEVELOPING A MONITORING PLAN AND SELECTING THE MONITORING PARAMETERS

This chapter discusses the key factors that influence the development of a monitoring plan and the selection of monitoring parameters. Probably the most crucial of these factors is the formulation of specific monitoring objectives, and this is the topic of Section 4.1. Section 4.2 defines the designated uses of water, and Table 2 qualitatively assesses the sensitivity of the different designated uses to changes in each of the monitoring parameters reviewed in these *Guidelines*. This is followed by an assessment of the effects of various management activities on each of the monitoring parameters (Section 4.3 and Table 3). Sections 4.3 and 4.4 evaluate the parameters with regard to the frequency of measurement and the cost of monitoring. The cost of monitoring is broken into separate categories, including the frequency of measurement, data or sample collection time, equipment costs, and analytic costs (Table 4). Box 5 (page 37) summarizes the ways in which the cost of a particular monitoring project might be reduced.

In many forested areas access can be a serious constraint to the frequency and location of sampling, and this is discussed in Section 4.6. The importance of existing data is reviewed in Section 4.7, and there is a brief discussion of how monitoring projects might evolve as a result of accumulating data. The final section of Chapter 4 briefly analyzes why it is so important to understand the physical features and processes of a watershed when designing a monitoring project.

4.1 PURPOSE OF MONITORING

The most important step in formulating a water quality monitoring project is the initial specification of the objectives. As discussed previously, the monitoring objectives often are the primary means for distinguishing among the seven different types of monitoring defined in Section 1.3. Identifying the objective(s) and type of monitoring then has

implications for the type, intensity, and scale of measurements (e.g., Table 1, page 8). Thus a very precise formulation of the monitoring objective(s) should lead to an efficient and effective water quality monitoring project. Vague or unrealistic objectives are likely to result in monitoring that collects unnecessary data and ultimately is unable to answer the pertinent management questions.

Careful formulation of the objectives is essential also because it precludes unrealistic expectations. Sometimes technical specialists will exaggerate the importance or capabilities of a water quality monitoring project in order to justify funding. However, water quality data often are ambiguous, and even the best statistical test carries a certain level of risk (Chapter 3). In the absence of a quick or definitive answer, the land manager or decision-maker can become disillusioned with the amount of resources required to sustain a water quality monitoring project, and this may result in the monitoring project being reduced in scope or even abolished. This not only inhibits the accumulation of long-term data but also reduces the credibility of the technical specialist. The resulting series of incomplete or discontinuous monitoring projects are self-defeating, as the data may not permit (1) the separation of management effects from natural variability; and (2) an understanding of the effects of rare events, such as a 5- or 10-year storm, on water quality and channel morphology.

The importance of properly formulating the monitoring objective can be illustrated by an example. A typical concern of forest managers, regulators, and fishery scientists is whether forest management activities are adversely affecting the fish in watershed X. This general question provides no indication as to whether the concern is directed towards trophy-sized trout or the biological integrity of the fish populations. A more specific identification of the designated uses and perceived adverse effect(s) is needed to determine whether the monitoring should focus on the number of fish, species diversity, total biomass, productivity, or condition.

Box 5: WAYS TO REDUCE THE COST OF A MONITORING PROJECT

I. INSTITUTIONAL STRATEGIES

1. What other agencies are currently collecting streamflow or water quality data (e.g., municipal water companies, fish hatcheries, hydropower facilities, National Park Service, etc.)? Cooperation with other agencies will reduce your costs.
2. What data have already been collected by other monitoring projects, or by other agencies or researchers? Historical data can be used to extend your records and help interpret your results. Water quality data from other projects or comparable sites can improve your sampling efficiency.
3. Who can help? Consider having members of the public conduct some of the water quality monitoring or sample collection. This may require additional quality control measures, but should help improve communication and understanding.
4. What are all your monitoring needs, and how might these be combined? Consider how to integrate different monitoring activities. For example, data from project monitoring also might be used for effectiveness, implementation, and baseline monitoring.
5. Can your objectives be made more specific? Narrower objectives will usually reduce the scope of a monitoring project by eliminating extraneous data.

II. METHODOLOGICAL STRATEGIES

6. Which inexpensive parameters can provide data relevant to the objectives? Some parameters are much less expensive in terms of their sampling frequency and cost of data collection and analysis. Can these provide the necessary data?
7. How much statistical rigor is required in the results? A reduction in the level of significance, for example, often can be justified in water quality studies, and this can reduce the frequency of sampling.
8. Is your sampling scheme efficient? Proper stratification of your sampling locations and sampling times can reduce the number of samples needed to characterize a given parameter with a specified level of uncertainty. A review of the monitoring project with a statistician can help ensure that your monitoring design is efficient and effective for your objectives.
9. Can a qualitative monitoring project achieve the primary goals? Often a visual inspection during critical periods can indicate whether a particular activity is generating adverse impacts. However, for this effort to be credible it may be necessary to involve representatives from the regulatory agencies or the concerned public.

eters most closely correlated with the limiting factor(s) for the fish population(s) of interest. The objective of the monitoring program might then be refined to a more specific question such as: Do the forest management activities in watershed X adversely affect the winter rearing habitat for coho salmon? This clarification of the monitoring objective immediately begins to suggest that certain habitat parameters, such as pool characteristics and large woody debris, might be more useful or easier to monitor than the actual population of over-wintering coho.

A further narrowing of the monitoring objectives can occur by identifying the specific management activities that could affect the designated uses of concern. In the above example, the question is what activities might reduce the availability of winter rearing habitat for coho salmon. If the road network in Watershed X is stable and well established, the monitoring might be directed towards evaluating the impact of forest harvest activities. In such cases information on the layout of the harvest units can be crucial to determining whether the impacts on riparian zones and stream channels will be direct or indirect. A harvest unit located well upslope, for example, would not be expected to directly affect

Under most circumstances a more precise set of objectives can be defined prior to initiating a monitoring project through discussions and qualitative investigations. If the general question is defined as the effects of forest harvest activities on the fish populations on watershed X, one might first identify the fish species present and the relative importance of the beneficial uses associated with those fish populations. Once the key species and beneficial uses have been identified, an experienced fish biologist should be able to qualitatively suggest what factor(s) might be limiting to the various species (e.g., spawning habitat, winter or summer rearing habitat, or food availability). This assessment then helps the manager to identify those monitoring param-

eters most closely correlated with the limiting factor(s) for the fish population(s) of interest. The amount or recruitment of large woody debris, but it could contribute sediment, which might reduce pool volumes. Harvest units adjacent to the stream channel could more directly influence the amount of cover and structure for over-wintering coho. Such spatial information can be very helpful in identifying which parameters should be included in a particular monitoring program (e.g., large woody debris or pool characteristics), which stream reaches should be monitored, and what type of upslope information needs to be collected in order to demonstrate a link between management activities and changes in water quality.

As noted earlier, the identification of the monitoring parameters has direct implications for the frequency and

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intensity of sampling. If, as in this example, both large woody debris and residual pool depth are likely to be affected by management activities and are limiting coho salmon populations, then these are the most appropriate monitoring parameters, and annual measurements are likely to suffice. On the other hand, if turbidity had been identified as a limiting factor, more frequent measurements over a range of flow conditions would be needed. Often a reduction in the anticipated frequency of measurements due to a shift in parameters will permit more sites to be sampled or allow other parameters to be monitored at each site.

This process of specifying the objectives usually will require more time and effort than simply initiating measurements of a standard water quality parameter such as turbidity or suspended sediment. Nevertheless, the potential savings in monitoring effort, and improvement in project results, usually makes this front-end investment extremely worthwhile.

Another example of a typical but unworkable objective is "determine the effect of recreational activities on water quality." Again this provides relatively little guidance regarding the parameters to be measured or the frequency and location of sampling. A more precise definition of the activities potentially affecting water quality is needed to develop an efficient monitoring project. After further discussion and investigation (e.g., the application of Tables 2-5 in these *Guidelines*), the objective might be refined to "determine the effect of overnight camping on the bacteriological quality of streams draining the XYZ Wilderness Area." This would yield a monitoring project that would focus on one or two of the bacterial parameters (Part II, Section 7.1), and measurements would be limited to a few sites during peak recreational use.

In some cases a clarification of the objectives might lead to the conclusion that a water quality monitoring project is not necessary. Implementation monitoring typically is an administrative review and does not involve water quality measurements. Effectiveness monitoring also may not require any inchannel measurements if it is evaluating a Best Management Practice (BMP), which is normally applied away from the stream channel. For example, if the monitoring objective is to determine if water bar spacing on skid trails is adequate to protect aquatic resources, the best approach would be to measure the sediment and runoff from a number of skid trails with a different spacing of water bars. Measurements of suspended sediment concentrations in the stream channel would have a much lower sensitivity because they integrate numerous other factors (e.g., bank stability, sediment storage in the channel, etc.) and are less sensitive to the management practice being evaluated.

A useful procedure to assess the adequacy of a proposed monitoring objective is to create a hypothetical data set consistent with the design of the monitoring project. If these hypothetical data can be analyzed in a way that meets the monitoring objective, then the monitoring plan can be

considered satisfactory. However, if the hypothetical data are insufficient or ambiguous, then it will be necessary to review the monitoring objective(s). The simple act of creating a data set and then directly relating the data to the objective(s) can be a very powerful tool for refining a monitoring plan.

In summary, clearly specifying the monitoring objective(s) is the single most important step in a developing a monitoring project. All too often the focus is on collecting data without due regard to the purpose for which the data is being collected. Often the monitoring parameters are selected because they are known and familiar, rather than because they are the most efficient or appropriate. Once a monitoring protocol is established, institutional inertia sometimes results in its continuation regardless of whether the monitoring objectives are being met.

4.2 DESIGNATED USES OF WATER

The rationale for public regulation of water quality is to protect the existing and designated uses of water (Section 1.4). Although the specific designated uses vary from state to state, they generally include agricultural use, industrial use, public water supplies, recreational use, and the propagation of fish and wildlife (EPA, 1988). Each state is responsible for determining which designated use(s) should be applied to the water bodies within that state. The designation of uses is important because it determines the water quality criteria that will be applied to that water body (EPA, 1988). Designation of coldwater fisheries as an existing or attainable use, for example, results in much more stringent water quality criteria than if industrial water supply were the only designated use. Establishing fish and wildlife propagation as a designated use is particularly useful in that it helps protect water bodies on the basis of their intrinsic value rather than relying solely on human uses, and it often leads to a large number of relatively stringent criteria. The general goal of the Clean Water Act of 1972—to support and propagate aquatic life—means that all states have a narrative water quality standard to protect the "biologic integrity" of the aquatic ecosystem. This, together with the anti-degradation policy, provides a relatively comprehensive framework for protecting water quality in forested areas (see Section 1.4).

Identifying the designated uses is another key step in developing a water quality monitoring project. Streams used for domestic water supplies, for example, probably should be monitored for bacteriological contamination, but this may be unnecessary if the only designated use is for the propagation of fish and wildlife. Conversely, pool parameters and bed material particle size may be very important if cold-water fisheries are a designated use, but are not nearly as relevant if the only designated use is for hydropower or public water supplies.

Table 2 presents a qualitative evaluation of the relationship between the monitoring parameters discussed in these

Table 2. 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.

Water quality parameters	Designated uses affected by water quality parameters						
	Domestic water supply	Agricultural water supply	Hydroelectric generation	Recreation	Warm-water fishes	Cold-water fishes	Biological integrity
<u>Water column</u>							
Temperature	3	4	4	2	3	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
Intergavel 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
<u>Flow</u>							
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
<u>Sediment</u>							
Suspended	1	1	1	2	2	2	1
Turbidity	1	2	1	1	1	1	1
Bedload	3	3	2	3	2	2	2
<u>Channel characteristics</u>							
Channel cross-sections	4	4	4	3	3	3	3
Channel 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
<u>Riparian</u>							
Riparian canopy opening	4	4	4	2	2	2	1
Riparian vegetation	4	4	4	2	2	2	1
<u>Aquatic organisms</u>							
Bacteria	1	3	4	1	4	4	3
Algae	2	3	4	1	2	2	1
Invertebrates	4	4	4	3	1	1	1
Fish	4	4	V	1	-	-	1

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Guidelines and the most common designated uses of water. These relative values cannot be assumed to apply under all conditions, but they provide an initial indication of which parameters are likely to be most directly related to that particular designated use most of the time.

The relationships suggested in Table 2 also do not mean that these parameters are the most appropriate for monitoring. A close relationship between a parameter and a designated use indicates that a particular parameter should be considered for inclusion in a monitoring project, but other factors must also be evaluated. Some of these other factors include the relative sensitivity of a parameter to both management activities and environmental factors; the type of management activities being carried out; the ease of measurement; the spatial and temporal variability of the parameter; the environmental setting; and the scale of the monitoring project. Although not all of these factors can be fully defined for each parameter, each of these is discussed in the following sections and in conjunction with the review of each parameter (Part II).

4.3 TYPE OF MANAGEMENT ACTIVITY

The type of management activity is another important consideration in developing a water quality monitoring plan. Each of the monitoring parameters discussed in these guidelines has a different sensitivity to human activities. Stream channel morphology, for example, is unlikely to be affected by forest fertilization, but may be relatively sensitive to grazing, road building, or road maintenance. Summer low flows might be increased by forest harvest, but are relatively insensitive to most other management activities except perhaps grazing.

Table 3 presents an empirical evaluation of the sensitivity of the monitoring parameters discussed in these *Guidelines* to a variety of management activities in forested areas in the Pacific Northwest and Alaska. The focus is on forest management activities, which are separated into forest harvest, road building and maintenance, forest fertilization, and the application of herbicides and pesticides. Other management activities in Table 3 include grazing, dispersed recreation, developed recreation/small communities, placer mining/sand and gravel extraction, and hardrock mining. Although these latter activities are outside the scope of this document, they have been included in Table 3 because they so often occur within the same watershed as forestry-related activities and their effect on streams must be considered when developing a water quality monitoring plan.

The values presented in Table 3 represent a qualitative, generalized assessment of the relative sensitivities of the parameters reviewed in Part II to individual management activities. Clearly there can be a great deal of variability with regard to the absolute impact of a given management activity, but the relative rankings should be consistent. The

suggested values also will vary according to a number of other factors, the most important of which probably is the environmental setting. Streams in a bedrock environment or a steep, V-shaped valley, for example, will be much less likely to experience changes in channel width because of grazing or road building than streams in an alluvial setting. The sensitivity of the parameters to a particular management activity also may vary according to stream size. These factors are discussed in more detail in Section 4.8.

The purpose of Table 3 is to help select those parameters worthy of further consideration in developing a water quality monitoring plan. An efficient monitoring plan should focus on those parameters that are most sensitive to past and planned management activities. Again, however, sensitivity to a particular management activity is not sufficient reason for inclusion in a monitoring project. One must also consider factors such as the costs of measuring the different parameters, and whether the natural spatial and temporal variability is likely to mask the effects of management. Road building activities, for example, are likely to affect peak flows, suspended sediment concentrations, and stream channel morphology, but these parameters vary greatly in terms of their ease of measurement and the time period needed to detect significant change. The following sections will identify and discuss these considerations in more detail.

4.4 FREQUENCY OF MONITORING

An important constraint in developing a monitoring plan is the anticipated cost of obtaining the necessary data. In this section the cost of acquiring data is analyzed in terms of the typical frequency of sampling and the range of flow conditions that need to be sampled. Section 4.5 considers the time required to obtain a sample, the equipment required to obtain a sample, and the cost of analyzing the sample or field data. All of these factors must be evaluated before one can estimate the cost of acquiring data on a particular monitoring parameter.

As discussed in Chapter 3, the sampling frequency is a function of the statistical objectives of the monitoring project. Any change in the desired accuracy or reliability of the results directly affects the sample size and the choice of parameters. All the parameters discussed in these *Guidelines* also are subject to spatial and temporal variability, and this again affects their relative precision and ability to detect change.

A monitoring project that is attempting to detect a relatively small change with a high degree of certainty will be more costly than a monitoring program with a lower standard for identifying a statistically-significant change. More measurements will increase the precision and hence the ability to detect change (Section 3.4.2), but the marginal cost and benefit of each additional measurement will vary according to the parameter.

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

Parameters	Sensitivity of monitoring parameters to management activity								
	Forest management activities						Other management activities		
	Harvest	Road building and maintenance	Applications			Placer mining ^a	Hardrock mining	Grazing	Recreation
		Fertilizers	Herbicides	Pesticides					
<u>Water column</u>									
Temperature	1-2	3	4	3	4	2	3	2	4
pH	3	3	3	3	4	3	1	3	4
Conductivity	3	3	3	3	4	3	1	3	4
Dissolved oxygen	3	3	2	3	4	3	3	1	4
Intergravel DO	2	2	3	3	4	2	2-3	2	3
Nitrogen	2	3	1	3	4	3	3	1	3
Phosphorus	2	3	1	3	4	3	3	1	3
Herbicides and pesticides	4	3-4	4	1	1	4	4	4	3
<u>Flow</u>									
Peak flows	1-2	1	4	3	4	4	3	3	3
Low flows	1	3	3	3	4	4	3	2	4
Water yield	1	3	3	3	4	4	3	3	4
<u>Sediment</u>									
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	3	2	4
<u>Channel characteristics</u>									
Channel cross-sections	2	1	4	3	4	1	3	1	4
Channel width/width-depth ratio	2	1	4	3	4	1	3	1	4
Pool parameters	2	1	4	3	4	1	3	2	4
Longitudinal or thalweg profile	2	1	4	3	4	1	3	2	4
Habitat units	2	1	4	4	4	1	3	2	4
Bed material									
Size	2	1	4	3	4	1	3	2	4
Embeddedness	2	1	4	3	4	1	3	2	4
Surface vs. subsurface	2	1	4	3	4	1	3	2	4
Large woody debris	1	4	4	3	3	3	4	4	4
Bank stability	2	1	3	2	4	2	3	1	3
<u>Riparian</u>									
Canopy opening	1-3	2	3	1	4	2	4	2	3
Vegetation	1-3	3	3	1	4	3	4	1	3
<u>Aquatic organisms</u>									
Bacteria	4	4	4	4	4	4	3	1	1
Algae	1	3	2	2	4	1	3	1	3
Invertebrates	1	1	3	3	2	1	2	1	3
Fish	2	1	3	3	3	2	3	2	1

^aPlacer mining also includes sand and gravel extraction.

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In the first column of Table 4, the monitoring parameters are grouped according to the typical frequency and timing of measurements. Parameters that need to be measured only annually, seasonally, or more frequently over a relatively short time period (e.g., daily for 2 weeks in mid-summer) are rated as having a low sampling frequency. These include most of the geomorphic and riparian parameters, as well as the forest chemicals such as herbicides and pesticides. Those parameters rated as having a high frequency of monitoring, such as the sediment parameters, must either be measured over all flow conditions or be intensively monitored over a series of high flow events.

The frequency of sampling for most of the water column parameters cannot be easily defined because of the large range of monitoring objectives. Low flow, baseline, or trend data might be obtained with relatively few measurements, while monitoring total nutrient loads (e.g., to protect downstream oligotrophic lakes) requires much more frequent sampling.

The second column in Table 4 indicates the flows over which sampling should be carried out in order to properly characterize the parameter of interest. For many parameters, such as those relating to channel characteristics and riparian conditions, the measurements can be made whenever it is practical and safe. Other parameters must be measured at high flows, and this can be an important constraint when access to the sampling site is difficult (Section 4.6), or when there is no bridge or other structure from which samples can be safely taken.

The channel and riparian parameters generally have the lowest measurement frequency and are the least restrictive with regard to the timing of measurements. This is due to the fact that they are—with the exception of habitat types—not flow-dependent. Although large discharge events can have a major effect, the parameters listed under channel characteristics usually are monitored on an annual basis.

In contrast, the three sediment parameters—turbidity, suspended sediment, and bedload—are highly dependent on discharge. Since virtually all of the sediment transport occurs during high flow events, and there can be considerable variation in sediment transport within a given storm event, frequent sampling is needed during high discharge events (Part II, Chapter 4). A similar logic applies to the monitoring of changes in water yield and the size of peak flows (Part II, Chapter 3).

Conductivity and the bacteriological parameters also are correlated with discharge, but they generally vary less than the sediment parameters. The relatively consistent inverse relationship between conductivity and discharge means that fewer samples are needed to determine conductivity as compared to bacterial concentrations. Bacterial contamination is more variable both within and among storm events.

The other biologic parameters—algae, invertebrates, and fish—exhibit seasonal variation. The optimal time and

frequency of sampling will vary with location and objective, but at a minimum the invertebrate populations should be sampled in the spring and fall (EPA, 1989b), and the resident fish population in winter and summer (Part II, Chapter 7).

The water column parameters exhibit considerable variation with regard to the desirable frequency of monitoring. Herbicides and pesticides are present only as a result of man's activities, and this simplifies the process of establishing a baseline or background level. The primary monitoring objective for herbicides and pesticides is to assess the inadvertent delivery of these chemicals into the aquatic ecosystem. Typically the highest concentrations occur immediately after application, and monitoring efforts are directed towards this relatively short time period. By predicting the average travel time from the application area to the monitoring site, an efficient sampling scheme can be developed. In most states only 4-8 samples are required to monitor an aerial application of herbicides or pesticides; the adequacy of this sample size is discussed in Part II, Section 2.6. The more persistent and mobile chemicals may have a secondary peak associated with the first runoff event, and this may require a second sampling period (similar to the sampling design for forest fertilization).

The frequency of sampling for nitrogen and phosphorus will depend upon the purpose of the monitoring and the type of management activities. A more intensive monitoring program may be required for forest fertilization than for herbicides and pesticides because there are several different pathways by which nitrogen and phosphorus might reach the stream channel. As was the case for herbicides and pesticides, there is an initial peak due to the direct application of fertilizer into the aquatic environment, and possibly a second peak in conjunction with the first runoff event following application. Detecting this second peak requires access to the sampling site on short notice, or frequent sampling using an automated sampler. The cost of monitoring for this second peak can be greatly reduced by randomly analyzing only a small proportion of all the samples collected between the initial peak and the first runoff event.

The sampling frequencies for dissolved oxygen, pH, and temperature are more complex because they often fluctuate both daily and seasonally. To obtain meaningful data, a monitoring project must either sample over this entire range or determine the most critical period and then consistently sample at this time. This means that an initial period of intensive sampling may be needed to determine the most sensitive period(s) for a particular parameter at a sampling site, after which the monitoring can be limited to that particular time.

4.5 COST OF MONITORING

Other key factors in assessing the cost of a monitoring project include the amount of staff time, funds, expertise,

Table 4. Frequency and cost of data or sample collection by monitoring parameters. L = low; M = medium; H = high; V = variable; NA = not applicable.

Parameter	Typical frequency	Flow conditions for sampling	Collection time	Equipment costs	Analysis costs
<u>Water column</u>					
Temperature	L-M	L	L	L	L
pH	L-M	L	L	L	L
Conductivity	M	All	L	L	L
Dissolved oxygen	L-M	L	L	L-M	L
Intergavel DO	M	V	L	M-H	L
Nitrogen	L-H	V	L	L	M
Phosphorus	L-H	V	L	L	M
Herbicides and pesticides	L	L-M	L	L	H
<u>Flow</u>					
Peak flows	H	H	M-H	M-H	H
Low flows	M	L	M-H	M-H	L-H
Water yield	H	All	M-H	H	H
<u>Sediment</u>					
Suspended	H	H	L-M	L	M
Turbidity	H	H	L	M	L
Bedload	H	H	M	M	M
<u>Channel characteristics</u>					
Channel cross-sections	L	L	M	M	M
Channel width/width-depth ratio	L	L	M	L	L
Pool parameters	L	L	M	L-M	L-M
Thalweg profile	L	L	M	M	M
Habitat units	L	L	M	L	M
Bed material					
Size	L	L	M	L	M
Embeddedness	L	L	H	L	L
Surface vs. subsurface	L	L	H	M	M-H
Large woody debris	L	L	M	L	L
Bank stability	L	L	L-M	L	L
<u>Riparian</u>					
Riparian canopy opening	L	NA	L-M	L-M	L-M
Riparian vegetation	L	NA	L-M	L	L
<u>Aquatic organisms</u>					
Bacteria	M-H	All	L	L	M
Algae	L-M	L	M	L-M	H
Invertebrates	L-M	L-M	L-M	L-M	M-H
Fish	L-H	L	H	M-H	M

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and equipment needed to make and interpret an individual measurement. The monitoring parameters evaluated in these *Guidelines* exhibit a wide variation in terms of their ease of measurement and in the equipment required. For many parameters a simultaneous discharge measurement is needed to properly interpret the data. Certain parameters also require more expertise to collect and analyze the field samples or data, and to interpret the results. Hence the value of a parameter for a particular monitoring project depends on the availability of staff time, expertise, equipment, and expendable funds for outside analyses.

The last three columns in Table 4 provide a qualitative ranking of the parameters with regard to the time needed to collect a sample, the equipment needed to collect a sample, and the costs of analyzing the sample or the raw data. For some parameters a range of techniques or measurements could be used, and the table is based on the techniques most commonly used for monitoring streams in forested areas. Associated costs, such as the need to house and maintain equipment, or improve access to the monitoring site, can be significant but are not included in Table 4 because these costs vary greatly.

For most of the physical and chemical constituents of water, the process of collecting the sample is relatively straightforward. However, for certain parameters, such as phosphorus, considerable care needs to be taken to avoid sample contamination. Equipment and analysis costs for the water column parameters vary from virtually nil in the case of temperature to more than \$100 per sample for a commercial analysis of pesticide or herbicide concentrations. The relative ease and low cost of measuring temperature, pH, and conductivity partly explains why these parameters are included in most monitoring programs.

Monitoring changes in water yield and the size of peak flows can be expensive because of the need to establish and maintain one or more stream gaging stations. A very long monitoring period will be needed to detect changes in the size of the peak flows with a long recurrence interval. In contrast, it may be possible to monitor low flows in a well-controlled reach without establishing a continuously recording gaging station.

The process of taking and analyzing suspended sediment and turbidity samples is not particularly difficult or expensive. Accurate bedload samples are more difficult to obtain. The main problems associated with measuring all three sediment parameters are (1) the need to sample intensively during high flows, (2) the need to simultaneously measure discharge, and (3) the difficulty of safely sampling small streams during high flow events. This last problem means that sampling locations are limited largely to bridges or other structures for all but the smallest (e.g., first-order) streams. Bedload sampling carries the additional problem of how to sample coarse (>5-10 cm) sediment.

Most of the channel morphology characteristics are more time-consuming to measure and analyze. Surveying

equipment is often required. As discussed in the previous section, a major advantage of these parameters is that they usually are measured on an annual basis.

The sampling frequency associated with the riparian parameters also is relatively low in most cases. Time and equipment needs can be characterized as low to moderate in most cases, although they can vary considerably depending upon the actual techniques being used.

Considerable variation can also occur in the time, equipment, and expertise needed to monitor algae, invertebrates, and fish. Generally the cost of obtaining a sample is moderate-to-high, and considerable expertise may be required to analyze samples of periphyton or invertebrates. The bacteriological parameters more closely resemble some of the water column parameters in that relatively little time is needed to collect the sample, but the analysis is moderately expensive. The sampling frequency may be relatively high depending upon the intensity of recreational use and the likelihood of contamination.

The differences in sampling and analytic costs shown in Table 4 are important in determining the spatial and temporal intensity of sampling and in selecting the appropriate parameter for monitoring. Many public land management agencies have seasonal staff who can be utilized for monitoring purposes, and a minimum of funds for outside analyses or equipment. This predisposes the monitoring program towards those parameters (e.g., channel characteristics) that can be measured annually and which do not incur high analytic costs. Private companies are often characterized as having fewer staff available for monitoring and more flexibility in contracting for outside analyses. These considerations must be recognized as a possible constraint to the development of an optimal water quality monitoring project.

4.6 ACCESS TO MONITORING SITES

The ease of access to a monitoring site, particularly during storm events, can be a controlling factor in selecting the parameters to be monitored. As shown in the second column of Table 4, several parameters must be measured during high flow events. If access to the sampling site is not possible during high flow events, or there are no structures from which measurements can be made, this precludes the use of those parameters. Many of the other parameters are relatively independent of discharge and can be measured at the most convenient time, such as during summer low flows. To the extent that a monitoring program is based on this latter category of parameters, access is not as important a criterion. However, ease of access can greatly affect the cost of a monitoring project, as transportation time is often the most expensive component.

Automatic water sampling devices may be able to alleviate the problem of access and sampling during high flow events, but they cannot eliminate it. Most automatic

samplers are unable to take more than 25-30 samples. Thus automated samplers can greatly reduce the number of trips needed to service a sampling site, but they cannot be left to take samples over an entire season. In most cases they are used to sample at specified time intervals, but particularly in smaller watersheds the sampling frequency may not adequately represent specific storm events. Intake conditions for automated samplers will vary during a storm event, and this can be a serious limitation to the accuracy of suspended sediment measurements (Part II, Chapter 4). Another potential problem with automated samplers is that their limited pumping capacity complicates their use in rivers or streams with more than a 15-ft change in water level or stage. Finally, automated samplers can only be used for parameters that are relatively stable over time. Hence they may be suitable for suspended sediment, but they cannot be used for parameters such as dissolved oxygen or pH.

As discussed in Part II, Chapter 4, a sampling scheme based on equal volumes of discharge can greatly improve the quality of the results. This requires coupling the automatic sampling device to a microprocessor and a continuously recording discharge measurement device (Thomas, 1985). Such sampling schemes are particularly helpful for calculating total fluxes of nutrients or sediment. Total flux data are most likely to be needed in the wasteload allocation process (Chapter 2) and for estimating the total nutrient loading for downstream areas.

4.7 AVAILABILITY OF EXISTING DATA

Another important consideration in developing a water quality monitoring plan is the amount of existing data. A major shortcoming of many monitoring projects is that they are initiated subsequent to management activities. This means the background or undisturbed value of a particular parameter must be extrapolated from a comparable, undisturbed site, which in many cases is difficult, and one can never completely resolve the question as to the comparability of the sites (Section 3.2.3).

The presence of an existing data set also is critical for putting an observed change in context. If a 50% change occurs in summer low flows or total flux of suspended sediment, is this a significant change that can be ascribed to a particular management activity? Determining a management-induced change requires one or more of the following: (1) a demonstration that it lies outside the normal range for that variable, (2) a demonstrable shift in values as compared to an adjacent basin, or (3) physical data that link a particular practice to an observed change in the stream channel. An analysis of change either at a particular site (e.g., trend monitoring) or in the relationship between treated and control sites (e.g., project monitoring) requires pre-disturbance data.

In cases when baseline or pre-disturbance data are inadequate, the third approach—collecting additional physi-

cal data to link management effects to changes in water quality—must be used. Often this will require measurements closer to the areas where the management activities are taking place (i.e., farther upstream or out of the stream channel). The intent is to directly observe the management-induced changes in erosion, runoff, or other processes, and relate these to changes in one or more of the instream parameters. Of course data making this linkage are useful even when pre-disturbance or long-term data sets are available, but they are particularly necessary when no data exist prior to management. As noted in the introduction (Section 1.1), upslope measurements and monitoring techniques are not discussed in these *Guidelines* even though some watershed information is essential to interpreting any instream water quality data.

As data accumulate there is an increasing capability to evaluate fluctuations and discern change. This can be a strong argument for continuing an existing monitoring project, even though the project may not necessarily be optimal in terms of parameter selection or sampling location. Often the best means to alter an existing project is to begin monitoring the additional desired sites or parameters while still continuing with the existing project. As data accumulate on the additional parameters or from additional sites, it may be possible to statistically relate these data to the original sites or parameters. Some of the parameters or sampling locations in the original monitoring project then can be eliminated. Clearly any alteration of a monitoring project will carry some cost in terms of the statistical reliability of the results, and this must be weighed against the potential benefits.

4.8 PHYSICAL ENVIRONMENT

4.8.1 ECOREGION CONCEPT

The physical environment must also be taken into account when selecting water quality monitoring parameters and designing a monitoring program. Forested areas in Washington, Oregon, Idaho, and Alaska exhibit considerable variability with regard to their climate, hydrology, geology, landforms, and soils. This variation is reflected in stream channel morphology, water chemistry, runoff patterns, aquatic flora and fauna, and the riparian ecosystem. This implies that the parameters discussed in these guidelines also will vary in terms of (1) the values that can be expected under undisturbed conditions, (2) their sensitivity to different management practices, and (3) their usefulness in detecting a change in water quality.

Resource managers have long attempted to group or classify areas with similar conditions. The purpose of classification is to justify the extrapolation of site-specific data to other areas, and to define a region for comparing data from different sites (e.g., identify strata for sampling).

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Numerous aquatic classification systems have been proposed, with each based on a specific set of physical characteristics and each having a somewhat different purpose (e.g., Bailey, 1976; Brussock et al., 1985; Hawkes, 1975; SCS, 1981; USGS, 1982). At present the ecoregion classification of Omernik (1987) is being applied by EPA to the continental U.S.

Ecoregions are defined as areas of relative homogeneity in ecological systems or in relationships between organisms and their environments (Crowley, 1967; Omernik and Gallant, 1986). Omernik (1987) identified 76 different ecoregions in the continental U.S. based on land surface form, potential natural vegetation, land use, and soils. The map by Omernik and Gallant (1986) indicates that 14

ecoregions are represented in Washington, Oregon, and Idaho. Eight of these are both extensive in area and have forests as their potential natural vegetation (Fig. 7). Ecoregions have not yet been defined or mapped for Alaska.

Some of the specific applications envisaged for the ecoregion concept include: (1) comparing similarities and differences within and among ecoregions; (2) helping to establish water quality standards in line with regional patterns of tolerance and resilience to human impacts; (3) helping to locate monitoring, demonstration, or reference sites; (4) facilitating extrapolation from site-specific studies; and (5) predicting the effects of changes in land use and pollution control efforts (Omernik and Gallant, 1986). These applications are largely untested although several studies

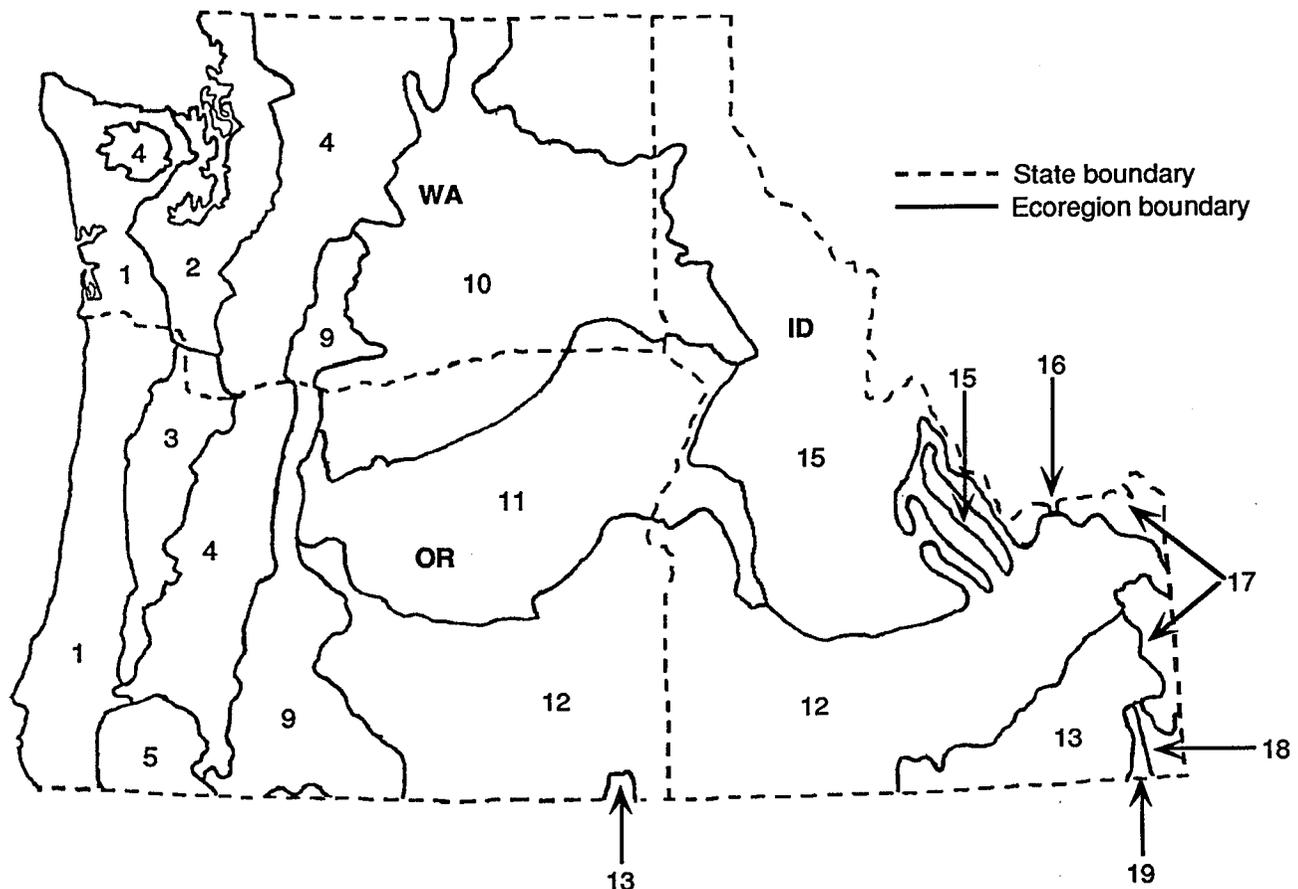


Figure 7. Ecoregions of Idaho, Oregon, and Washington. Numbers correspond to original numbers as designated by Omernik and Gallant, 1986. Asterisks indicate ecoregions with forests as their potential natural vegetation.

- * 1. Coast Range
- * 2. Puget Lowland
- * 3. Willamette Valley
- * 4. Cascades
- * 5. Sierra Nevada
- * 9. Eastern Cascades Slopes and Foothills
- 10. Columbia Basin
- * 11. Blue Mountains
- 12. Snake River Basin/High Desert
- 13. Northern Basin and Range
- * 15. Northern Rockies
- 16. Montana Valley and Foothill Prairies
- * 17. Middle Rockies
- 18. Wyoming Basin
- * 19. Wasatch and Uinta Mountains

have found a correspondence between ecoregions and spatial patterns in water chemistry and fish distributions in small streams in Arkansas and Ohio, respectively (Rohm et al., 1987; Larsen et al., 1986).

A recent study of 49 small streams in Oregon used statistical techniques to group physical habitat data, water quality data, and data on fish, invertebrate, and periphyton assemblages (Whittier et al., 1988). The greatest differences were between montane and nonmontane regions. Differences among the montane ecoregions were characterized as subtle. None of the data sets were able to distinguish all of the ecoregions included in the study, and the identified clusters were not consistent across data sets. Nevertheless, the results were regarded as providing support for the geographic classification of streams (Whittier et al., 1988).

4.8.2 CLIMATIC CONSIDERATIONS

Climate is a key driving force for runoff patterns and most of the erosion and sediment transport processes operating in a watershed. Different climatic regimes, when coupled with differences in factors such as the vegetation and geology, result in very different landscapes and landscape dynamics. This is the basic premise behind classification systems such as the ecoregion concept. It is important, however, not to rely on the mapped ecoregion boundaries as the basis for parameter selection and data comparisons, as ecoregions are generalized concepts which may not be applicable to specific sites. A preferred approach is to evaluate how the climatic regime affects the relative rates and importance of individual processes, and use this understanding to devise an appropriate water quality monitoring project.

Even though these *Guidelines* focus on monitoring parameters in the stream channel and riparian zone, a broader perspective is needed to interpret the data. The delivery of water and sediment to the stream channel is controlled by upslope processes, and climatic factors play a large role in determining the relative importance of the different processes. Climate plays an equally important role in defining some of the critical instream processes, such as the sediment transport capacity. Unfortunately direct relationships between climate and specific parameters, such as bed material particle size or width-depth ratios, usually cannot be drawn because of the influence of other factors such as stream gradient, stream size, and geology. Nevertheless, climatic conditions must be taken into account when designing a water quality monitoring project, and some of the more important considerations are discussed below.

First, the variability in precipitation is inversely proportional to average annual precipitation. Drier areas, such as eastern Oregon, eastern Washington, and southern Idaho, typically have more year-to-year variability in discharge, and for rain-dominated climates this implies that drier areas will have more temporal variability in sediment concen-

trations and sediment transport rates. Second, areas with more rainfall, such as the Olympic Peninsula, tend to have lower concentrations of nutrients and other dissolved ions. Third, higher discharges result in greater dilution, and this may make changes in concentration due to management activities more difficult to detect. In areas with a high annual discharge, a small change in nutrient concentration could result in a very different total flux, and this could be critical for downstream oligotrophic lakes.

The variation in the intensity and amount of precipitation is a key factor in determining the types and rates of sediment input into the stream channel. A variety of sediment production and delivery processes have been observed in the Western Cascades (e.g., Swanson et al., 1987), but there is little quantitative data on their relative importance. Mass failures are believed to be a dominant source of sediment for streams in steep, forested lands, and these usually are triggered by extreme storm events.

The type of precipitation can also be important. Forest harvest can significantly increase the size of the larger peak flows in areas subject to rain-on-snow events, and forest harvest can also increase the peak runoff rates during spring snowmelt (e.g., Troendle and King, 1985). In contrast, forest harvest in rain-dominated areas may not increase the larger peak flows (e.g., Wright et al., 1990). Hence knowledge of the climatic regime and the causes of flood events is essential to predicting the effect of forest management activities on peak flows of varying return intervals (Part II, Section 3.1). Such predictions are needed to determine if one should attempt to monitor changes in the size of peak flows.

A change in the size of peak flows also has implications for the stability of the banks, bed material, and large woody debris. Each of these channel features then has implications for the biotic component.

Other aspects of the climatic regime should influence the selection of monitoring parameters. In areas with a cool climate, for example, it may not be necessary to monitor changes in water temperatures due to forest harvest.

Climate also has a series of indirect effects on the various water quality monitoring parameters. These are too extensive to be detailed here, but they include effects on weathering rates, erosion rates and hence slope steepness, vegetation type, and productivity of the aquatic biota. The most important of these are discussed in the sections reviewing the individual parameters (Part II).

4.8.3 LAND FORM

Land form refers to the shape of the terrain and the pattern of the drainage system, and it is an important consideration when developing a monitoring project. In particular, the steepness of the sideslopes is a major factor in determining which runoff and erosion processes are

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likely to dominate, and in determining the rate at which water and sediment will be delivered to the stream channel. Stream gradient is one of the most important factors for determining the rate at which water, sediment, and large woody debris are moved downstream. Both the movement of material into the stream channel and the movement of material downstream directly affect monitoring parameters such as bank stability, channel morphology, turbidity, bed material particle size, and dissolved oxygen.

Areas with steep terrain generally will have more dynamic, high-energy streams. Steeper basins tend to produce more sediment (Swanson et al., 1987), with mass wasting being of primary importance. In these basins there typically is less of a lag between the production of sediment and its delivery into the stream system. These characteristics suggest that the steeper the basin, the greater the response to management activities (Swanson et al., 1987). This larger response to management activities does not necessarily make it easier to detect change through water quality monitoring. In steep basins the fine sediment will not be stored in the stream channel, and this could limit the ability of some of the channel parameters, such as embeddedness or bed material particle size, to indicate change. High gradient streams also are more likely to have cut down into bedrock, and this restricts the amount of change that could be expected in parameters such as thalweg profile, habitat types, and channel cross-sections.

Steep land forms generally are more susceptible to extreme events such as landslides and debris flows. Recent studies in the western Cascades suggest that sediment input and channel morphology often are dominated by relatively rare events (L. Benda, Univ. Washington, pers. comm.). If the basic pattern is one of severe disruption followed by a long period of recovery, it may be difficult for monitoring projects to detect the superimposed impact of management activities (e.g., Lisle, 1982). In areas of steep terrain it may be preferable to monitor upslope characteristics, such as the frequency of landslides in cut areas or along roads, rather than relying on instream measurements.

4.8.4 GEOLOGY AND SOILS

Geology is another important factor in determining landforms, stream characteristics, and soil types. The permeability, depth, and porosity of the soil and bedrock are critical to characterizing the runoff processes in a watershed. Soil and rock types affect the type of erosional processes and the rate of sediment delivery. Geologic features can control stream gradient and channel morphology.

An understanding of these geologic considerations is necessary for the proper design of monitoring projects. Bedrock channels are unlikely to show much change in

channel morphology or habitat types as a result of management activities. Turbidity may not be as useful a parameter in areas dominated by coarse-grained sediment. Conversely, bedload measurements may be difficult or relatively unimportant in streams with beds comprised of silt or other fine particles. Bedrock outcrops may control stream gradient in a particular reach, and this will reduce the sensitivity of the stream to deposition or erosion.

Background levels of nutrients and dissolved ions also are highly dependent on bedrock type and soil depth. The texture, depth, and permeability of the soil will influence the proportion of fertilizers, pesticides, and herbicides leached into the stream channel immediately after application and during the first runoff event. Coarse-grained soils tend to have higher permeabilities and less ability to capture and hold dissolved ions.

Soil texture is a major factor in assessing its susceptibility to compaction and surface erosion. Relative soil and bedrock permeabilities, among other factors, strongly influence hillslope drainage. High soil water contents and excess pore pressures are critical contributing factors to mass failures. Areas with frequent mass failures are difficult to monitor because these events tend to overwhelm the changes due to forest management activities.

4.8.5 SUMMARY

The physical factors discussed in the previous sections are only some of the more important considerations that must be taken into account when developing a water quality monitoring plan. The point is that one must be aware of the dominant physical and biological processes operating in the upslope areas and in the stream channel, and use this information to identify those parameters that are more likely to be affected by management and less likely to be subjected to extreme fluctuations by extraneous events. This knowledge is also necessary to relate changes observed in the stream channel to management activities.

Probability also plays a strong role in water quality monitoring. A monitoring plan implicitly assumes average conditions, and extreme events such as a 100-year flood, a large debris flow, or a volcanic eruption can disrupt even the most carefully designed monitoring project. The technical specialist designing the monitoring project must qualitatively assess the probabilities of different natural disturbances and structure the design of the project accordingly. As discussed in Chapters 2 and 3, any monitoring project must have (1) a procedure for regular data analysis and interpretation, and (2) flexibility to adapt as new information is acquired. If just these two components are actively incorporated into the monitoring project, then there is an excellent chance that the project will be successful.

5. PARAMETER RECOMMENDATIONS AND INTERACTIONS

5.1 RECOMMENDED PARAMETERS

The preceding chapters have shown that the selection and use of monitoring parameters depends upon a wide range of factors. The key factors were identified and discussed in Chapter 4, and the choice of parameters can be summarized by the notation:

Selected parameter(s) = function (objectives, designated uses, management activities, cost, and environmental setting).

Of these five controlling factors, the specific monitoring objectives were singled out as the most important factor in determining the parameters to be monitored. Specifying the objectives largely controls the frequency and location of sampling (Section 4.1), as well as the level of certainty and accuracy desired in the monitoring program (Chapter 3). It should be noted that the level of certainty and accuracy specified in the objectives also affects both the choice of parameters and the cost and location of sampling; this is just one example of the important interactions among the five key factors identified above.

The sensitivity of the designated uses of water to changes in the water quality parameters is discussed in Section 4.2 and qualitatively evaluated in Table 2. Table 3 ranks the sensitivity of the water quality monitoring parameters reviewed in these *Guidelines* to the most prominent human activities in forested areas in the Pacific Northwest and Alaska. The intent of these two tables is to provide an initial screen for identifying those parameters most important for protecting a specific designated use (Table 2), and those parameters most likely to be affected by a particular management activity (Table 3).

Other key considerations in developing a water quality monitoring program are monitoring costs, the availability of existing data, access to the monitoring sites, and the physical

environment the monitoring site. Table 4 provides a general rating of both the sampling frequency needed to evaluate each parameter, and the flow conditions under which a particular parameter typically needs to be sampled. The second part of Table 4 indicates the relative cost of measuring a parameter, and this was broken down into three components: the time needed to collect a sample or field data, the equipment needed to collect a sample or field data, and the cost of analysis. Again this information is intended to serve as a screen or filter for selecting the most appropriate monitoring parameter for a particular situation.

Some of the factors that affect the selection of monitoring parameters cannot be summarized in tabular form. The physical environment affects both the spatial and temporal variability of a parameter, and the sensitivity of different parameters to specific management activities (Section 4.8). Hence consideration of the physical environment must be done on a case-by-case basis, and this requires a sensitivity to, and knowledge of, watershed processes.

The importance of these case-specific factors means that these *Guidelines* cannot specify which monitoring parameters are most appropriate under all conditions. Nevertheless a combination of Tables 2-4 and the distilled experience of those involved in this project provides a qualitative indication as to the monitoring parameters most likely to be useful most of the time.

Table 5 integrates all this information into a qualitative ranking of the usefulness of each parameter evaluated in these *Guidelines* to monitor the water quality effects commonly associated with various management activities. For the purposes of this table "usefulness" is based on the (1) sensitivity of a parameter to the specified management activity; (2) importance of a parameter with regard to the designated uses characteristic of forested areas in the Pacific Northwest and Alaska; and (3) typical costs of measurement and data analysis, including consideration of the sampling frequency and time needed to detect change. A quantitative

Table 5—cont.

Parameters	Usefulness of parameters to monitor the effects of management activities										
	Forest management activities					Other management activities					
	Harvest	Road building and maintenance	Fertilizers	Herbicides	Pesticides	Grazing	Dispersed recreation	Developed recreation/rural populations	Placer mining ^a	Hardrock mining	
<u>Channel characteristics—cont.</u>											
Habitat units	3	3	4	4	4	2	4	3	2	4	
Bed material particle Size	2	2	4	4	4	2	4	3	2	4	
Embeddedness	3	2	4	4	4	2	4	3	3	4	
Surface vs. subsurface	3	2	4	4	4	3	4	4	3	4	
Large woody debris	2	3	4	4	4	4	4	3	4	4	
Bank stability	2	2	4	3	4	1	3	3	3	4	
<u>Riparian</u>											
Canopy opening	2	2	4	2	4	2	4	3	3	4	
Vegetation	2	2	4	2	4	1	2	3	3	4	
<u>Aquatic organisms</u>											
Bacteria	4	4	4	4	4	1	1	1	4	4	
Algae	3	4	2	2	4	2	3	3	3	3	
Invertebrates	2	2	3	3	1	2	3	2	2	1	
Fish	3	3	4	3	3	2	4	4	3	1	

^aPlacer mining also includes sand or gravel extraction.

^bIn almost all cases, discharge data should be collected concurrently with these parameters.

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procedure was not used in preparing Table 5 because each factor integrates several sub-factors, and not all the sub-factors are amenable to a numerical ranking. In the following sections the results and rationale are summarized for each of the major management activities listed in Table 5.

5.1.1 FOREST HARVEST

The cutting and yarding of trees affects streamflow and runoff patterns, disturbs the soil and exposes it to erosion; it also can decrease slope stability and alter the inputs of organic material and light into the stream system. This range of potential effects suggests that numerous parameters could be used to monitor the water quality impacts of forest harvest. However, each potential parameter has a unique set of advantages and disadvantages.

As discussed in Chapter 2 of Part II, forest harvest is known to alter water chemistry, but these changes generally are not large enough to limit the designated uses of water. A further disadvantage of monitoring water chemistry is that observations need to be made over a range of flow conditions. Intergravel dissolved oxygen (DO) is likely to be one of the most useful of the chemical and physical components of water if fine sediment is a concern.

A large number of paired-watershed experiments have shown that forest harvest usually increases total water yield, increases the size of the smaller peak flows, and increases summer low flows. Detecting these hydrologic changes requires several years of data and—with the exception of summer low flows—considerable effort. Assuming that large areas of the catchment are not compacted (e.g., no more than 15%), the potentially most significant change is in the size of peak flows in areas subject to rain-on-snow events. However, change in the size of the larger peak flows (e.g., events greater than the mean annual flood) are precisely those changes that are most difficult to detect because of the need to measure and compare infrequent peak flows.

Absolute changes in the rate of sediment transport are difficult to measure because of the need to intensively sample high flow events and the difficulty of obtaining accurate results. This makes trend or validation monitoring a difficult objective. Suspended sediment and turbidity monitoring may be more successful if done on a comparative rather than absolute basis. A typical example is a comparison of turbidity levels upstream and downstream of a particular activity (e.g., project monitoring). To be statistically valid, such comparisons must be replicated and include either a pre-disturbance calibration period or a comparison to other paired sites that are not treated (Section 3.2). Intergravel DO may be a very useful surrogate for fine sediment as it is often easier to measure.

Most channel characteristics are easier to monitor because measurements typically are made only on an annual basis. They also can be directly related to one of the most important and restrictive designated uses, namely habitat

quality for coldwater fisheries. These channel parameters have two primary limitations: (1) They can be highly variable within a given reach, and (2) their relative response to forest harvest and natural events in different environments is still difficult to predict and distinguish. The case study from the Little North Fork of the Clearwater River in Idaho indicates that cobble embeddedness was selected as the best parameter for monitoring the effects of forest harvest and fire on fish habitat (Box 6).

Stream temperature and the riparian vegetation can be very useful monitoring parameters if forest harvest extends into the riparian zone. Both of these are relatively sensitive and easy to measure. Evaluation of the riparian canopy opening using a RAPID-type technique (Grant, 1988) can be very useful to quickly assess current condition over a large area, but it is not advocated as a monitoring technique because of its lower sensitivity and the lag period between management activities and observed change (Part II, Section 6.1).

Of the biologic parameters, the macroinvertebrate community probably offers the greatest promise for monitoring. Periphyton also might be effective for monitoring purposes, but samples are more difficult to collect and analyze. Certain fish species can be very sensitive to forest harvest activities, but difficulties in measurement techniques and the presence of confounding factors may make it difficult to directly link management activities to the particular fish parameter(s) being monitored.

In summary, temperature and riparian vegetation are likely to be among the most useful monitoring parameters if the forest harvest activities are near enough to the stream channel to affect stream shading and the input of organic materials. The usefulness of turbidity or suspended sediment will depend on the objectives of the monitoring. Intergravel DO and some of the channel characteristics should be considered as indirect indicators of changes in upslope erosion, sediment transport, and runoff. Invertebrate monitoring can serve as the means to link the physical changes (temperature, turbidity, and channel morphology) to the biological integrity and designated uses of the stream. Less frequent measurements of large woody debris are useful to evaluate trends in this component of fish habitat and to help assess the adequacy of the silvicultural prescriptions to maintain the input of large woody debris to streams.

In general the effects of forest harvest are proportional to the percent of the area disturbed and the percent of the vegetation removed. Other factors that can ameliorate or exacerbate the effects on water quality include the yarding system employed (e.g., skyline, tractor, or helicopter), the location of the harvest units with regard to ephemeral and permanent stream channels, the pattern of harvest within the catchment, the sensitivity to mass failures, and the particular climatic conditions following harvest.

Clearcutting normally has a more severe impact than selection cutting per unit area, as all the trees are being removed. However, selection cutting has to disturb a larger

BOX 6. CASE STUDY: SEDIMENT MONITORING IN THE LITTLE NORTH FORK OF THE CLEARWATER RIVER, IDAHO

The Little North Fork of the Clearwater River is a 30-mile long watershed of mixed federal, state, and private forest lands in northern Idaho. Although much of the watershed has been subjected to extensive road building and logging over the past several years, some sub-drainages are still pristine roadless areas.

Fish species in the Little North fork are predominantly west slope cutthroat trout and native rainbow trout, with some Dolly Varden and eastern brook trout. During the fall kokanee trout from Dworshak Reservoir spawn in the Little North Fork and its tributaries. Though fish population data are scarce in this region, there has been growing concern that populations are decreasing and species composition is shifting to more sediment-tolerant species.

Much of the watershed has a high erosion hazard due to steep slopes, unstable granitic soils, and a tendency for mass-wasting. The two primary geologic materials in the basin are the Idaho Batholith (a large igneous intrusion) and a Pre-Cambrian metamorphic mica schist.

Before this project little information was available on the sediment conditions of streams in the watershed. Some streams appear to have large quantities of sediment, but there is no documentation of stream sediment conditions prior to the recent management activities. The Idaho Division of Environmental Quality (DEQ) saw a need to determine baseline (i.e., pristine) sediment levels for the tributaries and main stem of the Little North Fork of the Clearwater River. This baseline data was to be complemented by an evaluation of sediment quantities in various tributary watersheds with differing amounts of erosive soils, burned areas, and forest management activities.

Cobble embeddedness was chosen as the best parameter to quantify instream sediment. During 1988 and 1989, cobble embeddedness was measured in 22 stream reaches, and the average number of hoop samples per reach was 18. Measured cobble embeddedness varied from 22-93%.

In cooperation with DEQ, the Idaho Department of Fish and Game began inventorying habitat conditions and fish populations in the Little North Fork and its tributaries in 1990. The resulting data will be used to evaluate the relationship between fish and sediment as measured by cobble embeddedness and habitat conditions.

Idaho DEQ is now inventorying the percentage of erosive soils, acres harvested, miles of road, and burned acres within each drainage. These factors then will be related to the cobble embeddedness data. In future years trends in cobble embeddedness will be monitored throughout the Little North Fork Clearwater River.

Source: Jack Skille, Division of Environmental Quality, Idaho Department of Health and Welfare, Coeur d'Alene, Idaho.

area in order to obtain an equivalent volume of wood. Commercial and precommercial thinning generally have the least impact. The similar effects of these different forest harvest operations suggest that the parameter recommendations apply equally well to clearcutting, selective harvest, and thinning, even though the relative magnitude of the water quality impacts may differ.

Site preparation is not explicitly considered in Table 5 because it usually follows immediately after forest harvest, and its water quality impacts are both highly variable and difficult to separate from the water quality impacts of forest harvest. Extensive brush raking or other mechanical soil disturbance can result in site prep having a more detrimental effect on water quality than the actual cutting and removal of trees. Because site prep usually has a greater effect on soil disturbance than vegetation removal, sediment-related parameters generally will be of primary importance.

5.1.2 ROAD BUILDING AND MAINTENANCE

The primary concern associated with road building and maintenance is the increased rate of erosion. Often this is best monitored in the smaller ephemeral channels that directly

drain the road prism because the relative effects are much greater than in the higher-order downstream channels. Measurements of erosion and sedimentation can be direct (e.g., suspended sediment, thalweg profile) or indirect (e.g., intergravel DO). Other potential water quality problems due to road building and maintenance include higher conductivities attributed to road salting, the runoff of fertilizer and herbicide residues from cut and fill slopes, and the input of sand applied to improve winter traction into stream systems.

The selection of parameters to monitor sediment from road building and maintenance presents the same Hobson's choice as described for monitoring sediment from forest harvest activities. The channel morphology parameters represent simple techniques that are still being proven. Monitoring of turbidity and suspended sediment concentrations requires relatively intensive storm sampling. Intergravel DO may not provide a direct link to management actions. The riparian parameters are more appropriate for evaluating long-term change than short-term water quality effects. Aquatic macroinvertebrates probably are the best choice among the biological parameters, but they may not be as sensitive as some of the physical parameters.

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These considerations suggest that a project to monitor the water quality effects of road construction and road maintenance might best rely on some of the channel characteristics and direct observations on or adjacent to the road network during storms. With regard to the individual channel characteristics, sediment deposition might be monitored by measuring residual pool depth, pool volumes, or longitudinal profiles. A change in the balance between sediment inputs and sediment transport could be inferred by monitoring changes in the bed material particle size. Measurement of deposition and scour, and changes in the bed material particle size, should provide a reasonable assessment of whether and how adverse effects might be occurring. Turbidity or suspended sediment observations on the water draining from the road network during storm periods could provide direct evidence for any observed changes in the larger stream channels. Monitoring aquatic invertebrates might provide useful supplemental data and help establish the link between sediment and aquatic organisms.

5.1.3 FOREST FERTILIZATION

Selecting parameters to monitor the application of forest chemicals is much simpler because of the very specific impact of the chemicals on water quality. Most forest fertilization programs in the Pacific Northwest apply only organic nitrogen or urea (Gessel et al., 1979). Monitoring the effects of forest fertilization on water quality is best achieved by taking water samples after the application of fertilizers and then during the first runoff event following fertilization. These samples should be analyzed for the major forms of nitrogen. Discharge data is required if concern exists over the total flux of nitrogen as well as the maximum concentration. Total flux is essential when there is a desire to minimize eutrophication. Simultaneous temperature measurements may be helpful as an indicator of the rate of biological activity during the period of increased nutrient availability due to forest fertilization. Suspended sediment data may be useful because adsorbed nutrients can be a substantial component of the total nutrient budget.

The indirect and biological effects of fertilization can be best monitored by measuring the algal community (e.g., biomass, chlorophyll-a, or growth). In slower-flowing streams it may be possible to detect substantial changes in the algal community by measuring daily fluctuations in pH.

5.1.4 APPLICATION OF HERBICIDES AND PESTICIDES

The most efficient and direct approach for monitoring the water quality effects of herbicide and pesticide applications is by taking and analyzing water samples. Although the analytic costs may be relatively high, in most cases only a few samples need to be taken immediately after application.

The potentially high analytic costs can be reduced in several ways. One approach is to first analyze a composite sample, and then analyze individual samples only if the first test indicates significant contamination. A second approach is to mix a dye tracer with the chemical and use this to indicate which samples should be analyzed. Finally, trace enrichment cartridges can be used to estimate the total flux over the period of sampling (NCASI, 1984). For the more persistent and mobile chemicals, additional sampling during the first runoff event may be necessary.

Again temperature measurements may be helpful as an indicator of chemical reaction rates and biological activity during the period of monitoring. Discharge measurements are needed if total losses to the aquatic system are being estimated. Suspended sediment data are necessary if the adsorbed component is of concern.

For herbicides, an alternative approach is to monitor changes in the canopy cover or some other aspect of the riparian vegetation. Generally such measurements will be a less sensitive indicator because the riparian vegetation will respond only to acute doses of herbicides. On the other hand, monitoring the riparian vegetation avoids the problem of capturing a transient peak concentration in the stream, and changes in the riparian vegetation can be directly linked to several designated uses.

Similarly the effect of pesticide applications on water quality can be monitored on a coarse scale by sampling the aquatic invertebrate populations. A disadvantage of using the riparian vegetation to monitor herbicides, and aquatic invertebrates to monitor pesticides, is the ubiquitous problem that an observed change may be due to other factors.

5.1.5 GRAZING

Of all the management activities considered in Table 5, grazing has the widest range of water quality effects. Grazing can affect water quality by changing the pattern and timing of runoff, and increase sediment loads by removing the vegetative cover and trampling the streambanks. Animal wastes can directly impair water quality through bacterial contamination and increasing nutrient levels. This range of effects means that almost all of the parameters discussed in these *Guidelines* could be used for monitoring grazing impacts on water quality.

The choice of monitoring parameters in a particular situation will depend on the designated uses as well as the intensity and pattern of grazing. Bacterial contamination is important if domestic water supply or recreation is a designated use. Nutrients will be more critical if eutrophication of downstream water bodies is a concern. Bank stability is less likely to be a useful monitoring parameter if the riparian areas are fenced off or if there are sufficient watering points away from the stream channel.

In general, however, livestock tend to congregate in riparian areas. In these cases bank stability and the riparian

vegetation are two parameters that can be quickly assessed and are directly affected. These two parameters may not necessarily be the most sensitive to grazing effects over an entire watershed, but they are often among the first to be affected because grazing tends to be concentrated in the riparian zone. The other channel characteristic that might be particularly useful to monitor grazing impacts is the channel cross-section, as this can provide data on bank slope, channel aggradation, and stream widening.

Either phosphorus or nitrogen can be used to monitor the additional nutrient inputs due to grazing. Phosphorus may be preferred to nitrogen as it generally is more responsive to grazing, and the background levels of phosphorus in forested areas typically are very low. Compliance monitoring can be relatively simple and inexpensive because accompanying discharge data are not needed, and sampling can be directed towards those times when concentrations are most likely to exceed the water quality criteria (e.g., summer low flows). If the total nutrient load is of concern, monitoring costs will be greatly increased because sampling will have to be done over the entire range of flows, and continuous discharge data will be required.

In the absence of other management activities, it might be possible to evaluate the effects of increased nutrient levels by monitoring algae or chlorophyll-a. Use of either of these parameters presumes that the algae are nutrient-limited, which may not be the case in shaded, forested streams (Part II, Section 6.2).

5.1.6 DISPERSED RECREATION

The primary means by which dispersed recreation can adversely affect water quality is through the inadequate disposal of human wastes. Since in most cases the recreational users are using the same water for drinking and perhaps bathing, bacteriological monitoring usually will be of primary importance.

The inadequate disposal of human waste also will increase nutrient inputs, but in most streams this effect will be relatively minor and difficult to detect. However, nutrients can accumulate in lakes, and regular monitoring of phosphorus and nitrogen may be needed to evaluate the impact of dispersed recreation on oligotrophic lakes, particularly in the alpine zone. The focus of recreational use on lake shores and stream banks means that in heavily used or fragile areas, riparian vegetation and bank condition should be monitored.

5.1.7 DEVELOPED RECREATION/RURAL POPULATIONS

Water quality impacts of developed recreation sites, such as ski resorts, and rural populations involve a mix of activities that lie largely outside the scope of this document. Some of the concerns, such as the input of nutrients and the use of herbicides and pesticides, are analogous to the forest

management activities considered earlier. Other concerns, such as the quality of runoff from streets and other paved areas, are not included in these *Guidelines* and are not consistent with the monitoring parameters reviewed here. Nevertheless, Table 5 does identify those parameters most likely to be useful for monitoring some of the water quality impacts expected from developed recreation areas and rural settlements.

For example, the potential pollution due to wastewater and septic tanks is best monitored by bacteriological and nutrient parameters. Storm sampling and discharge measurements generally will be necessary because elevated levels of bacteria and nutrients are most likely to occur during storm events. Specific conductance might be useful to indicate the amount of wastewater being discharged if the conductance of the receiving waters is both known and relatively low.

Local use of herbicides and pesticides can be expected, but the scale of application suggests that neither of these is likely to significantly impair water quality. The irregular use of such chemicals suggests that tissue analysis or sediment sampling may be a better method to estimate herbicide and pesticide loadings than the sporadic and costly analysis of water samples.

Almost any rural settlement and associated human activities will result in an increased sediment load. The need for monitoring this effect should be based on a field assessment of the potential for increased erosion and consideration of the downstream designated uses. If there is a need for sediment monitoring, it usually will be analogous to project monitoring and involve a comparison of measurements upstream and downstream of the settlement or recreation site.

Rural settlements and developed recreation sites often result in a modification of the stream channel by localized clearing of the riparian vegetation and the construction of roads, culverts, etc. This leads to a variety of effects on stream temperature, channel morphology, and many other parameters. Again the monitoring of such effects should be done only after the need is clearly identified through a qualitative evaluation of water quality and stream condition, and with due consideration for the designated uses.

5.1.8 PLACER MINING/SAND AND GRAVEL EXTRACTION

These activities are outside the scope of this document, but they have been included because they sometimes occur in forested environments in the Pacific Northwest and Alaska, and they can confound the effects of forest management activities on water quality. The primary concerns with regard to placer mining are the release of fine sediment and the destabilization of the stream channel. Direct monitoring includes the three sediment parameters (turbidity, suspended sediment, and bedload), as well as many of the

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channel characteristics. Indirect monitoring can be done with other parameters such as intergravel DO or some of the biological parameters—particularly invertebrates or coldwater fishes. Often turbidity and suspended sediment data will complement rather than duplicate data on the channel characteristics. The precise channel characteristic of greatest sensitivity and utility will depend on the size of the sediment being released and the transport capacity of the stream. Generally a combination of bed material particle size and an aggradation indicator (e.g., pool depth, channel cross-sections, or longitudinal profile) is likely to provide the most useful information that can be directly linked to placer mining and the designated uses. Changes in channel morphology can trigger secondary effects on the size of the riparian canopy opening and the riparian vegetation. Although these latter two parameters may be useful for broad-scale assessments, generally they will be less useful for monitoring because they are secondary effects and therefore less sensitive to change.

The release of suspended sediment is also a concern for sand and gravel extraction, and this suggests that similar monitoring guidelines should apply. Since removal of large amounts of sand and gravel may destabilize the stream channel by altering the sediment load, some monitoring of the channel characteristics is essential. The precise parameters to be monitored will depend on factors such as the stream gradient and the extent to which the channel is constrained or incised.

5.1.9 HARDROCK MINING

Again this activity is outside the general scope of these *Guidelines*, but a brief discussion is pertinent because hardrock mining often occurs in forested areas. The effects of hardrock mining on water quality may also confound or complicate water quality monitoring projects focusing on forest management activities. Although historic mining activities often have resulted in excessive sediment inputs, hardrock mining should not alter the sediment input or channel morphology provided proper management practices are used. Typically the greatest impact of current hardrock mining activities is on stream chemistry. The precise parameters that will be most affected depend on the type of mine, the chemical characteristics of the rock being mined, and the extraction process being used. This uncertainty is acknowledged in Table 5, but generally it is important to monitor at least pH and conductivity. Any change in these parameters should then trigger a more extensive evaluation of stream water chemistry.

Since most water chemistry parameters are difficult to continuously monitor, it is important to regularly sample aquatic macroinvertebrates or other organisms with a life span of at least several months. This monitoring will ensure detection of sudden, toxic releases that might otherwise not be detected by periodic water sampling. Flow measure-

ments may also be necessary to complement the chemical data and ensure that the mining operation is not withdrawing excessive amounts of water.

5.1.10 WILDFIRE AND PRESCRIBED BURNING

Wildfires and prescribed burning were not included in Table 5 because their effects on water quality are both diverse and variable. Removal of the vegetative canopy will reduce transpiration and tend to increase water yield, especially during the growing season and immediately afterwards. Loss of the canopy cover along streams can increase peak summer water temperatures (Wright, 1978) and make temperature an important monitoring parameter. Generally both nitrogen and phosphorus inputs into the aquatic system will increase after a fire, and this is due primarily to the disruption of the terrestrial nutrient cycles (Tiedemann et al., 1978; Wright, 1978).

The effects of fire on sediment yield vary according to the frequency and intensity of fires, the steepness of the hillslope and drainage network, and the extent to which the vegetation controls the movement and storage of sediment (Swanson, 1978). Fire can greatly increase surface erosion by temporarily creating a hydrophobic soil layer (e.g., Dyrness, 1976; Megahan and Molitor, 1975). Fire also increases surface erosion and sediment delivery rates by removing the litter layer and organic debris that traps sediment both on hillslopes and in the stream channel. The magnitude of these effects will depend on the geomorphic sensitivity of the landscape, and this is largely a function of slope steepness (Swanson, 1978).

These data suggest that nitrogen and phosphorus should each be monitored when downstream eutrophication is a concern. In such cases continuous discharge data must also be collected. The need to monitor stream temperature will depend on the extent to which the riparian vegetation was removed by fire. In steep lands substantial increases in sediment yield can be expected, and the selection of monitoring parameters will depend largely on which erosional processes were altered by the fire. An increase in surface erosion may transport only the fine particles (e.g., sand-sized or smaller), and inchannel measurements should focus on turbidity or possibly suspended sediment. If coarser materials are being delivered to the stream channel by debris flows or other mass movements, one may wish to monitor some of the channel characteristics such as channel cross-section, width-depth ratios, or thalweg profiles. Complementary data on the amount of large woody debris in the stream channel should be collected if the fire burned into the riparian zone (the zone of future recruitment) or consumed some of the large woody debris within the active channel.

The Silver Fire case study (Box 7) is one example of how a broad range of monitoring activities were integrated

Box 7. CASE STUDY: SILVER FIRE RECOVERY PROJECT,
SISKIYOU NATIONAL FOREST

In 1987 the second largest fire in Oregon's history burned 96,500 acres of the Siskiyou National Forest. Approximately 56% of the burned area, or 53,600 acres, was located within the Kalmiopsis Wilderness. In the non-wilderness portion, trees containing an estimated 262 million board feet of lumber were killed by the fire and were potentially available for salvage. However, most of this area had no road access, and prior to the fire the environmental community had actively sought designation of the roadless areas as wilderness or national park.

The burned area also included more than 30 miles of the Illinois River. This is one of the most remote whitewater rivers in the continental U.S., and it is designated as a Wild and Scenic River. Under the antidegradation policy (Section 1.4), no deterioration of water quality can occur that will interfere with, or be injurious to, the designated uses.

Following the recommendations of an interdisciplinary project team, nearly a dozen timber salvage sales and related recovery projects were conducted over a 2-year period. The recovery projects were aimed at (1) salvaging as much of the burned timber as possible; (2) minimizing erosion from the fire and harvest activities; (3) protecting the designated uses, including salmon and steelhead habitat; and (4) restoring wildlife habitat. Specific project activities included 11 miles of new road construction, helicopter and limited cable harvest of fire-killed trees, reforestation of all harvested and sensitive areas within 2 miles of a road, aerial seeding of about 6,000 acres of the most intensely burned area with a mixture of annual grasses, installing checkdams on sensitive drainages, contour felling of trees on intensely burned sites, and installing fish habitat improvement structures in selected areas using native materials.

Concern over the effects of the salvage sales and the effectiveness of the recovery projects resulted in a series of monitoring activities. These were developed with the assistance of various governmental agencies, researchers, and public groups. Monitoring activities were developed to answer four basic questions: (1) were the protective measures implemented as planned; (2) were the recovery projects and mitigation efforts effective in preventing additional damage to the resources in the burned area; (3) were the assumptions used to predict the effects of the fire and subsequent timber salvage valid; and (4) what is the status of the designated uses of water after timber harvest and fire recovery?

Fourteen of the 30 monitoring activities focus on water quality, fish habitat, and other water-related resources, and these are summarized in the table. Specific objectives for the monitoring activities were based on the designated uses considered to be at risk. For example, steelhead and salmon fisheries were identified as important designated uses; pool and summer rearing habitat were identified as the most important limiting factors to these uses. Thus the monitoring efforts are directed towards parameters such as maximum summer water temperatures, pool volumes, stream shading, and fish cover. In some of the more productive reaches, a series of cross-sections are being used to assess changes in channel aggradation/degradation, bed material particle size, and habitat types. Landslides were identified as the largest potential source of sediment, and the number, size, and location of major landslides are being monitored by the periodic analysis of aerial photos.

Sampling locations were established at the mouth of the main tributaries entering the Illinois River and on the Illinois River itself. Sampling locations along the Illinois River were selected according to availability of access and proximity to both key fish habitat areas and salvage activities. Sampling times were selected to maximize the likelihood of detecting management-induced effects. As indicated in the table, the duration and frequency of sampling varies by monitoring activity.

Monitoring efforts within the Silver Fire Recovery Project were designed to balance instream measurements related to water quality and the designated uses with upslope implementation and effectiveness monitoring. In general, the individual monitoring activities will not be able to demonstrate the effects of specific management activities on the various designated uses. However, the combined data should be able to assess BMP effectiveness and identify important changes in the designated uses and their respective cause(s). Annual reports will summarize the data collected, evaluate conditions in the burned area, and assess the effects of management activities. This information will be available to other governmental agencies and the public.

Source: Siskiyou National Forest, P.O. Box 440, Grants Pass, Oregon 97526.

Box 7—cont. Summary of the water quality monitoring activities within the Silver Fire Recovery Project.

Monitoring activity	Type of monitoring*			Cost by year in thousands of \$ (Year 1 is first year after fire)										Total	
	IM	E	V	IN	1	2	3	4	5	6	7	8	9		10
Group 1															
Steelhead and chinook populations and habitat quality	-	X	X	X	24	21	21	-	-	21	-	-	-	21	108
Determine turbidity levels and principal nonpoint sources	-	X	-	-	20	5	5	5	5	5	5	5	5	5	65
Continuous monitoring of stream water temperatures	-	X	X	X	17	2	2	-	2	-	2	-	-	2	27
Group 2															
Verify log suspension and protection of streamside management unit in cable-logged areas	X	-	-	-	1	1	1	-	-	-	-	-	-	-	3
Determine whether 100-ft buffers along streams were maintained during logging	X	-	-	-	3	-	-	-	-	-	-	-	-	-	3
Assess implementation of BMPs with regard to roads and landings	X	-	-	-	2	2	-	-	-	-	-	-	-	-	4
Group 3															
Erosion and deposition in Class III and IV streams	-	-	X	-	3	-	2	-	2	-	-	2	-	-	9
Particle-size and cross-section monitoring on selected tributaries	-	-	X	-	4	2	2	-	2	-	2	-	-	2	14
Assess blowdown and bank stability within stream buffer zones ¹	-	X	X	-	2	1	1	-	-	-	-	-	-	1	5
Growth and survival of planted trees in riparian areas	-	X	-	-	1	-	1	-	-	1	-	-	-	1	4
Evaluate changes in low flows, size of peak flows, and infiltration rates on two tributary watersheds	-	-	X	X	8	-	2	-	-	2	-	-	-	2	14
Erosion from roads and resulting sedimentation of streams	-	X	-	-	2	-	-	-	-	-	-	-	-	-	2
Landslide inventory	-	-	X	X	3	-	-	8	-	-	8	-	-	8	27
TOTAL (thousands of \$)					90	34	37	13	11	29	17	7	5	42	285

*Type: IM = implementation, E = effectiveness, V = validation, IN = inventory.

¹Plus 1 additional year after a large blowdown event.

to address a variety of management concerns and objectives. This case study is particularly interesting because it explicitly spelled out the different frequencies and duration of each component of the monitoring program.

5.2 EXPERT SYSTEM

Table 5 and the accompanying discussion provide one simple means for assessing which parameter(s) might be most useful for a particular monitoring project. However, more than one management activity may need to be monitored, more than one designated use may be adversely affected, or more time, funds, or equipment may be available than is implicitly assumed in Table 5. Table 5 also assumes that measurements could be made during high flow events, and this assumption was made to keep Table 5 from becoming too unwieldy.

To overcome these limitations and make better use of the information contained in the *Guidelines*, an interactive expert system has been developed. This is called PASSSFA (Parameter Selection System for Streams in Forested Areas), and it uses a relatively inexpensive, commercially-available expert system shell called VP-Expert (Paperback Software, 1989).

In functional terms, an expert system is a computer program that reasons about a problem in much the same way, and with about the same performance, as specialists (Waterman, 1986). Typically expert systems are designed to help people think through difficult problems and provide suggestions about what to do without taking over every aspect of the task. The ultimate goal is to allow people knowledgeable in the subject area, but with less training and experience than an expert, to achieve nearly the same level of performance as an expert with regard to a certain set of decisions. Unlike traditional computer programs that use set algorithms, expert systems use rules based on reasoning or judgment (Buchanan, 1989). Expert systems can and do incorporate model simulations, but these simply provide additional input to the decision-making process.

PASSSFA relies on over 300 rules that were developed from the information contained in Chapter 4 and Section 5.1. These rules specify the response of parameters or groups of parameters to the controlling factors (management activities, designated uses, frequency of sampling, time required to collect a sample, equipment costs, analysis costs, and access during high flows). From the user's response to these questions, PASSSFA generates a list of suggested monitoring parameters. PASSSFA runs on IBM or IBM-compatible personal computers with at least 384K of RAM and DOS version 2.xx or later. PASSSFA uses video display screens with CGA, EGA, VGA, or a Hercules monochrome adapter. A color monitor is preferred.

Running the program does not require a user's guide, as the first four screens provide the necessary introductory

information. The program then prompts the user with a series of questions, with the number of questions varying according to the responses received. When sufficient information has been obtained to select from among all the parameters reviewed in these *Guidelines*, those parameters that meet the user's needs and criteria are displayed along with a confidence factor.

The confidence factors indicate on a scale of 0 to 100 the relative certainty that a particular parameter will be useful. The values assigned are based on experience and judgment, and they have no statistical meaning and cannot be quantitatively analyzed. Because PASSSFA selects only those parameters that meet the specified criteria, the confidence factors associated with the parameters displayed at the end of a consultation range from a minimum of 65 to a maximum of 100. In most cases the estimated confidence factors do not exceed 85 or 90, as one is rarely certain that a particular parameter will be useful under all possible conditions. It also is possible that no parameter will meet the user's criteria; in this case the final screen displays the message "None available given constraints." The user must then relax one of the criteria (e.g., indicating other designated uses for the water body in question, increasing the amount of time or funds available for monitoring, or allowing access during high flows). A "What if" function in the program allows the user to modify one response without repeating the entire consultation, and this is particularly useful when the initial result is a null set of parameters.

The advantages of referring to PASSSFA in conjunction with these *Guidelines* are as follows: (1) PASSSFA lists an estimated confidence factor with each recommended parameter, (2) multiple management activities and designated uses can be evaluated at one time, (3) the explicit inclusion of a range of possible responses in the rule base yields a set of recommended parameters more directly tailored to the user's needs and responsibilities, and (4) the effects of a change in one controlling factor can quickly be evaluated. Some of the disadvantages include the following: (1) an inability to easily print out the results of a particular consultation, and (2) the generation of unrealistically high confidence factors when more than one management activity or designated use is selected. Both of these disadvantages result from the particular software shell used to construct PASSSFA.

The problem of generating unrealistically high confidence factors appears only when more than one designated use or management activity is selected. In such cases a parameter may be selected by more than one rule. Although parameters selected by several rules will be listed only once at the end of the consultation, the confidence value associated with that parameter is altered. Specifically, the confidence factor is calculated by adding the confidence factors of the individual rules that are satisfied (expressed as decimals rather than on a scale of 0 to 100), and then subtracting the product (again calculated using decimal values). For ex-

Part I

ample, if the same parameter was selected by two different rules with confidence factors of 70 and 80 (0.70 and 0.80), respectively, the combined confidence factor in decimals would be $0.70 + 0.80 - (0.80 \times 0.70)$, or 0.94. In PASSSFA this parameter would then be displayed at the end of the consultation with a confidence factor of 94.

In the case of water quality monitoring, the simple selection of a parameter by two different rules does not necessarily mean that particular parameter is more likely to be useful. Local conditions and professional judgment still have to be applied. More realistic estimates of the confidence factor can be obtained only by repeating the consultation with only one management activity and one designated use, and recording the confidence values obtained in each case. The "what if" function of the expert system shell facilitates this type of repeated consultation.

Cost considerations precluded the inclusion of PASSSFA with each copy of the *Guidelines*, but copies of the expert system can be obtained by sending a blank, formatted diskette with at least 225K-bytes of available space to the Seattle office of the U.S. Environmental Protection Agency at:

U.S. EPA, Region 10
NPS Section, WD-139
1200 Sixth Ave.
Seattle, WA 98101

5.3 PARAMETER SELECTION AND INTERACTIONS

The discussion in Section 5.1 illustrates several important points regarding the selection of water quality monitoring parameters for forest management activities. First, in most cases the choice of monitoring parameters is not easy or clear-cut. Road building and maintenance and forest harvest are two common management activities where the most direct and sensitive water quality monitoring parameters are also the most difficult and costly to measure. The choice of parameters is much clearer with regard to monitoring fertilizer, herbicide, and pesticide applications, but these activities are less frequent and generally do not pose a chronic threat to the designated uses of water. The absence of well-defined monitoring parameters for forest management activities should not be surprising since the uncertainty regarding what to monitor was a principal rationale for the preparation of these *Guidelines*.

A second general conclusion that emerges from Table 5 is that the traditional physical and chemical water column parameters have limited usefulness for monitoring most management activities in forested areas. As indicated in Table 3, the primary water quality effects of road building and maintenance and forest harvest stem from the increased

sediment load and the reduction in the riparian vegetation. These changes can adversely impact most of the usual physical and chemical water quality constituents, but with the exception of temperature these effects are generally small or indirect. Suspended sediment and turbidity are two physical water column parameters that can be used to directly monitor changes in the amount of fine sediment, but in most cases intensive monitoring is needed during storm events in order to obtain useful data.

Third, the parameters listed under channel characteristics should be considered in any monitoring project having changes in sediment, flow, or riparian vegetation as a possible concern. These parameters have the advantage of being easier to measure and integrating the effects of all the individual storm events. Their primary disadvantages are (1) the lack of long-term data to evaluate their usefulness for water quality monitoring, and (2) the difficulty of relating observed changes to specific management activities.

Fourth, one or more of the biologic parameters is ranked as at least moderately useful as a monitoring technique for almost all of the various management activities. The advantage of the biologic parameters is that they can be directly related to the designated uses of water, they often are quite sensitive to management impacts, the sampling frequency is low-to-moderate, and the cost of sampling and data analysis also can be considered moderate. It may not always be clear what specific biological parameter should be monitored, but EPA's current emphasis on developing techniques and criteria for biological monitoring appears to be well founded. The danger is that some of these techniques will be adopted before they can be fully validated for the different ecoregions.

These four generalizations are less applicable to the other management activities of mining, grazing and recreation. The most appropriate parameters for monitoring sand and gravel extraction, placer mining, and recreation (either dispersed or developed) are relatively clear-cut. It is more difficult to generalize about the choice of parameters for monitoring hardrock mining because there is so much variation in the extraction and processing of the ore.

The potential impacts of grazing on stream systems are so extensive that a wide variety of water quality parameters could be utilized. Like grazing, wildfires can affect a variety of stream and channel parameters, and the selection of parameters will depend on the physical environment, the designated uses, and the intensity and location of the fire relative to the stream channel.

An important limitation of Table 5 is that each parameter is considered independently. However, many of the parameters are closely related. Turbidity, for example, is often used as a surrogate for suspended sediment concentration. Channel cross-sections, width-depth ratios, thalweg profiles, and pool parameters all are responsive to changes in the balance between discharge and sediment concentrations. These types of interrelationships mean that not all parameters that are rated highly in Table 5 should be used.

The best approach is to identify the parameters of most interest from Tables 2-5, and then rely on the technical review of the individual parameters in Part II to help make the final choice.

Tables 6A and 6B provide one way to overcome the problem of considering each parameter independently. These qualitatively rank the effect of change in one parameter on all the other parameters reviewed in these *Guidelines*. The intent of these tables is that once one possible monitoring parameter has been identified, the tables can be used to assess which other parameters are most likely to respond to changes in that parameter. Closely related parameters can be identified, and this information should help to eliminate any duplication of monitoring effort and maximize the independence of the parameters selected for monitoring.

Proper use of the tables requires careful attention to the cause-and-effect ordering of the interactions. For example, the width-depth ratio can greatly affect water temperature, but water temperature has only a very tenuous effect on the width-depth ratio.

A second limitation of Table 5 is that it does not explicitly link the various parameters that might be combined to produce an optimal monitoring program. Bed material particle size and thalweg profile, for example, might be a powerful combination to evaluate sediment-related changes in alluvial channels. The particular combination most appropriate for a given situation can be determined only through an understanding of how management activities affect the various processes operating in a watershed. Hence the *Guidelines* cannot provide specific solutions, but the combination of Tables 5, 6, and Part II should allow an informed decision to be made.

Similarly, Table 5 does not necessarily indicate which combination of parameters is best suited to demonstrate the effect of a particular management activity on the designated uses of a water body. In the absence of a well-replicated study, or associated data on the causal processes, a simple observation of trends for the one or two "most useful" parameters will not rigorously demonstrate the cause of an observed trend. Table 6 does provide some indication of cause-and-effect linkages by indicating how one parameter will affect each of the other monitoring parameters reviewed in these *Guidelines*. However, if water quality is to

improve, monitoring results must feed back into management practices, and this can only be done effectively when the cause of an observed change in water quality can be identified.

Determining cause-and-effect often is further complicated by the presence of multiple management activities upstream of the monitoring site. For this reason it is important to understand why a particular parameter is being recommended in Table 5 and how it is, or is not, linked both to the management activity and the designated uses. This information is explicitly set forth in the discussion of each parameter in Part II, and both the manager and the monitoring specialist must be aware of the strength of these linkages.

There is also an inherent limitation as to what can be learned from monitoring activities that are limited to the stream channel and adjacent riparian areas. Inchannel data are essential for determining trends in water quality, evaluating whether the designated uses are being impaired, and assessing whether the applicable water quality standards are being met. For practical reasons these *Guidelines* had to focus on those parameters and monitoring techniques that can be utilized in the stream channel and adjacent riparian zone. However, a full understanding and interpretation of these data requires a broader watershed perspective. Knowledge of the type and location of management activities and upslope processes is necessary if the monitoring results are to provide effective feedback to management and enhance our understanding of watershed behavior.

Ultimately it is the value of this feedback to management that determines whether a particular monitoring project is successful. Similarly, the internal feedback loops largely determine whether the data collection and analysis efforts within a particular monitoring project will meet the project objectives. At least for the foreseeable future, the development and operation of water quality monitoring projects will remain an iterative process. The range of controlling factors and natural conditions simply precludes the development of an optimal water quality monitoring project on the first try. These *Guidelines* are one means to facilitate or advance this iterative process for water quality monitoring projects in forested areas.

Table 6A. Interrelationships among water quality monitoring parameters. Table to be read as the effect of the parameter at the top of each column on the parameters listed on the left-hand side of the tables: 1 = highly sensitive, directly related; 2 = moderately related (may be indirect); 3 = weakly or multiple-step related; 4 = largely unrelated.

Parameters	Water column						Changes in flow			Sediment					
	Temp.	pH	Conduct.	DO	IDO ^a	N	P	Herbicides	Pesticides	Peak	Low	Discharge	Suspended	Turbidity	Bedload
<u>Water column</u>															
Temperature	—	4	4	4	4	4	4	3	4	4	2	2	3	3	2
pH	1	—	3	3	3	3	3	4	4	4	3	3	3	3	4
Conductivity	1	1	—	3	3	3	3	4	4	1	1	1	1	1	4
Dissolved oxygen	1	3	4	—	2	2	2	4	4	4	1	2	2	2	4
Intergravel DO	2	3	4	1	—	2	2	4	4	4	1	2	2	3	2
Nitrogen	3 ^a	2 ^a	3	3 ^b	3	—	3	3	4	4	3	3	2	2	4
Phosphorus	3 ^a	2 ^a	3	3 ^b	3	3	—	3	4	4	3	3	2	2	4
Herbicides	3	3	3	3	3	4	4	—	4	4	4	2	3	3	4
Pesticides	3	3	3	3	3	4	4	4	—	4	4	2	3	3	4
<u>Flow</u>															
Peak flows	4	4	4	4	4	4	4	4	4	—	4	—	3	3	4
Low flows	4	4	4	4	4	4	4	3	4	4	—	—	4	4	2
Water yield	4	4	4	4	4	4	4	3	4	1	1	—	4	4	4
<u>Sediment</u>															
Suspended	4	4	4	4	4	4	4	3	4	1	3	1	—	1	3
Turbidity	4	3	4	4	4	4	4	3	4	1	3	1	1	—	3
Bedload	4	4	4	4	4	4	4	3	4	1	4	1	3	4	—
<u>Channel characteristics</u>															
Channel cross-sections	4	4	4	4	4	4	4	3	4	1	4	2	3	4	1
Channel width/width-depth ratio	4	4	4	4	4	4	4	3	4	1	4	1	3	4	1
Pool parameters	4	4	4	4	4	4	4	3	4	2	4	1	3	4	1
Thalweg profile	4	4	4	4	4	4	4	3	4	2	4	4	3	4	1
Habitat units	4	4	4	4	4	4	4	4	4	2	3	1	3	4	1

Table 6A—cont.

Parameters	Water column					Changes in flow				Sediment					
	Temp.	pH	Conduct.	DO	IDO ^a	N	P	Herbicides	Pesticides	Peak	Low	Discharge	Suspended	Turbidity	Bedload
<u>Channel characteristics—cont.</u>															
Bed material															
Size	4	4	4	4	4	4	4	3	4	1	3	1	3	4	1
Embeddedness	4	4	4	4	4	4	4	3	4	1	3	1	3	4	1
Surface vs. subsurface	4	4	4	4	4	4	4	4	4	1	3	1	3	4	1
Large woody debris	4	4	4	4	4	4	4	3	4	2	4	3	4	4	4
Bank stability	4	4	4	4	4	4	4	3	4	1	4	1	3	4	1
<u>Riparian</u>															
Canopy opening	4	4	4	4	4	4	4	3	4	1	4	1	3	4	1
Vegetation	4	4	4	3	3	3	3	3	4	2	3	1	4	4	1
<u>Aquatic organisms</u>															
Bacteria	1-2	3	4	2	2	1	1	4	4	2	2	1	2	2	4
Algae	2	3	4	2	3	1	1	1	4	3	2	2	2	2	1
Invertebrates	1-2	2	4	1	1	2	2	2	1	3	2	2	1	2	1
Fish	1-2	2	4	1	1	2	2	3	3	2-3	1-3	2	1	1	1

^aIntergravel dissolved oxygen.

^bAffects speciation rather than total concentration.

Table 6B. Continuation of Table 6A for channel characteristics, riparian, and aquatic organisms. Table to be read as the effect of the parameter at the top of each column on the parameters listed on the left-hand side of the tables: 1 = highly sensitive, directly related; 2 = moderately related (may be indirect); 3 = weakly or multiple-step related; 4 = largely unrelated.

Parameters	Channel characteristics										Riparian		Aquatic organisms			
	Channel cross-sec.	Chan. width/ width-depth	Pools	Thalweg profile	Habitat units	Bed size	Embed- edness	Surface vs subsurface	LWD	Bank stability	Canopy opening	Vege- tation	Bacteria	Algae	Invertebrates	Fish
<u>Water column</u>																
Temperature	1	1	1	2	3	4	4	4	3	3	1	4	4	3	4	4
pH	3	2	3	4	4	3	3	4	3	4	4	3	4	1	4	4
Conductivity	4	4	4	4	4	4	4	4	2	4	3	4	4	3	4	4
DO	2	2	2	3	4	3	4	4	3	3	3	3	3	1	3	3
Intergavel DO	2	2	2	3	4	1	1	1	4	3	3	3	3	1	3	3
Nitrogen	4	3	4	4	4	4	4	4	4	4	3	3	4	2	4	4
Phosphorus	4	3	4	4	4	4	4	4	4	4	3	3	4	2	4	4
Herbicides	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4
Pesticides	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4
<u>Flow</u>																
Peak flows	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4
Low flows	4	4	4	4	4	4	4	4	3	3	2	4	4	4	4	4
Water yield	4	4	4	4	4	4	4	4	4	4	3	4	4	4	4	4
<u>Sediment</u>																
Suspended	4	4	4	4	4	1	3	4	1	3	2	4	4	4	4	4
Turbidity	4	4	4	4	4	1	4	4	1	3	2	4	4	4	4	4
Bedload	3	2	3	3	3	1	3	4	1	3	3	4	4	4	4	4
<u>Channel characteristics</u>																
Channel cross- sections	—	1	1	3	3	3	4	4	1	3	2	4	4	4	4	4
Channel width/ width-depth ratio	1	—	1	3	2	3	4	4	1	3	3	4	4	4	4	4
Pool parameters	2	2	—	2	1	3	4	4	1	3	3	4	4	4	4	4
Thalweg profile	1	2	1	—	3	3	4	4	1	3	3	4	4	4	4	4
Habitat units	2	2	1	2	—	3	4	4	1	3	3	4	4	4	4	4

Table 6B—cont.

Parameters	Channel characteristics										Riparian			Aquatic organisms		
	Channel cross-sec.	Chan. width/width-depth	Pools	Thalweg profile	Habitat units	Bed size	Embed- edness	Surface vs subsurface	LWD	Bank stability	Canopy opening	Vege- tation	Bacteria	Algae	Invertebrates	Fish
Bed material																
Size	2	2	1	2	2	—	1	1	1	4	3	4	4	4	3	
Embeddedness	2	2	2	2	2	1	—	1	2	4	3	4	4	4	3	
Surface vs. subsurface	2	2	2	2	2	1	1	—	2	4	3	4	4	4	3	
Large woody debris	3	3	4	4	3	4	4	4	—	2	1	4	4	4	4	
Bank stability	2	1	2	3	3	1	4	4	1	3	1	4	4	4	4	
<u>Riparian</u>																
Canopy opening	1	1	1	4	4	4	4	4	3	—	1	4	4	4	4	
Vegetation	1	1	3	4	4	4	4	4	2	1	—	4	4	4	4	
<u>Aquatic organisms</u>																
Bacteria	4	4	3	4	3	4	3	4	4	3	3	—	4	3	4	
Algae	2	2	3	3	3	2	1	2	4	1	2	3	—	1	3	
Invertebrates	2	2	2	3	1	1	1	1	2	1	1	1	1	—	2	
Fish	1	1	1	1	1	1	1	1	1	2	1	2	1	1	—	

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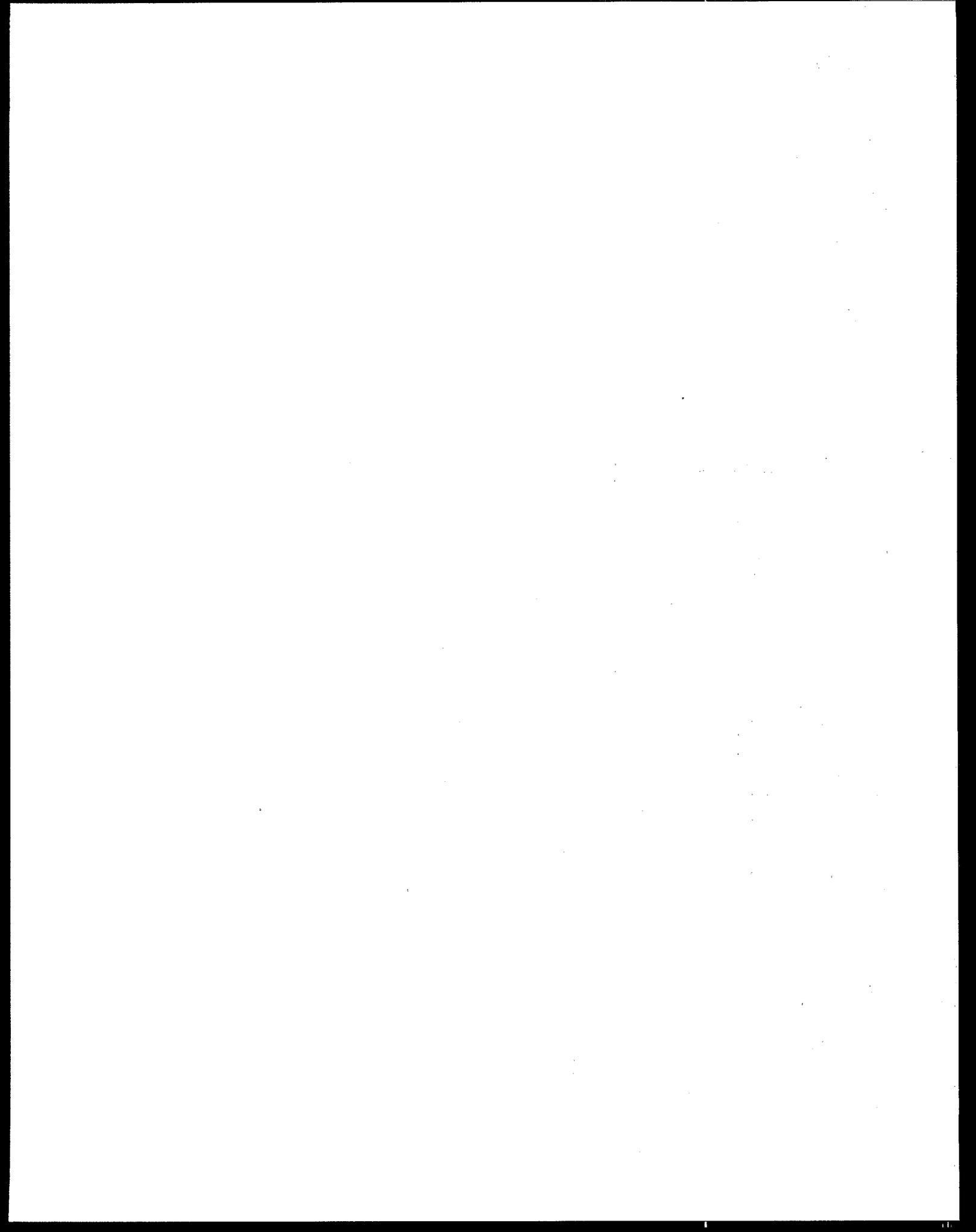
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PART II

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

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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:

$$^{\circ}\text{C} = 5/9 (^{\circ}\text{F} - 32)$$

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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

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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:

$$\text{pH} = \log 1/[\text{H}^+] = -\log [\text{H}^+]$$

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 (CO₂). Compared to most other atmospheric gases, carbon dioxide is relatively soluble, and in solution it combines with water to form carbonic acid (H₂CO₃). The equilibrium between carbonic acid and its component ions (H⁺ and HCO₃⁻) 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, CO₂-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 CO₂ 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 μ S) 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 CO₂ 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-

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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 conduc-

tivity 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 $\mu\text{mhos/cm}$, where $10^6 \mu\text{mhos}$ is equal to 1 mhos. Conductivity may also be reported in millisiemens/meter, with 1 millisiemen/m equal to $0.1 \mu\text{mhos/cm}$ (APHA, 1976).

Pure water not in contact with the atmosphere has a conductivity of approximately $0.05 \mu\text{mhos/cm}$. Normal distilled or deionized water has a conductivity of at least $1.0 \mu\text{mho/cm}$, 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 $42 \mu\text{mhos/cm}$ (Hem, 1970). The range for potable water in the U.S. is 30 to $1500 \mu\text{mhos/cm}$. 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 $\mu\text{mhos/cm}$ 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.

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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 ground-

water 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-

¹The 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). Generally 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.

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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

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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 organic 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 N₂ 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 $0\text{-}30^\circ\text{C}$ and a pH range of $6.5\text{-}9.0$ (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 $<0.3 \text{ mg L}^{-1}$ 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.

Temperature ($^\circ\text{C}$)	pH		
	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

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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 suggest that relatively intensive monitoring for urea, total ammonia, and nitrate for approximately 4 days usually will detect the peak nitrogen concentration due to the direct application of fertilizer into the fluvial system. Simultaneous discharge data should also be collected to permit an estimate of the total amount of fertilizer delivered directly into the stream. High losses suggest a need to modify the method and conditions of fertilizer applications.

Monitoring plans should recognize that a second increase in nitrate concentrations often occurs during the first runoff events following fertilization. Although this latter peak also is unlikely to violate water quality standards, the higher flows mean that the majority of the nitrogen loss can occur at this time (Norris et al., 1983). This second set of samples will indicate the overall ability of the terrestrial nitrogen cycle to scavenge and fix additional nitrogen inputs.

The reality of operational forest fertilization programs is that proper application procedures—specifically the use of pellets and buffer strips—generally should prevent aquatic nitrogen levels from exceeding national standards (e.g., Cline, 1973). Often the increase in aquatic nitrogen will be large in relative terms, but small in absolute terms. This increase may enhance aquatic productivity for a couple of years without any apparent adverse impacts to fast-flowing streams. More careful monitoring is needed when the water is used for domestic consumption within a relatively short distance from the fertilized area, and when slow-flushing ponds or lakes are present downstream. Generally the monitoring station(s) should be as close to the application area as possible, as this will minimize both the dilution effect and biotic uptake.

Mandatory monitoring and stricter water quality criteria may be appropriate for basins that drain into oligotrophic lakes. In these cases there is a long-term, cumulative hazard to lake water quality. This danger is best addressed by setting stricter standards for fertilizer applications and minimizing erosion.

Very intensive monitoring may be required in those basins designated as impaired under Section 319 of the Clean Water Act. In these basins a water quality model and a load allocation process may need to be developed for the water quality constituents limiting the designated uses of water (Part I, Section 1.4). An assimilative capacity also must be determined and allocated between point and non-point sources (Ice, 1990). If nitrogen is one of the constituents of concern, the monitoring program will have to evaluate the timing and frequency of standard violations, the relative contribution of each source, natural background levels, the

validity of the water quality model, and changes due to management actions.

Assessment

Almost any forest management activity will affect some aspect of the nitrogen cycle. Logging, fire, and forest fertilization can substantially increase nitrogen concentrations in streams. However, in most cases the absolute amount of nitrogen which enters into streams is small because most forest streams have such low background levels of nitrogen compounds. The observed increases in aquatic nitrogen caused by logging and fire typically represent only a very small fraction of the total nitrogen capital on site (e.g., Sollins and McCorison, 1981).

The forest management activity with the greatest potential for increasing nitrogen concentrations in forest streams is forest fertilization. Aerial applications of nitrogenous fertilizers such as urea generally deliver some nitrogen directly into the drainage system. When efforts are made to limit drift and avoid direct application into streams and lakes, water quality standards should not be exceeded (e.g., Cline, 1973; Fredriksen et al., 1975; Bisson, 1982). This suggests that a limited amount of water quality monitoring should suffice in most cases.

In evaluating the effects of an increase in nitrogen concentrations in streams, one must consider all the transformations and processes of the aquatic nitrogen cycle. For example, an increase in organic nitrogen will not affect primary productivity as much as an increase in inorganic nitrogen. In some streams an increase in plant-available nitrogen will have little biologic effect because primary productivity is limited by other factors. In many cases a small or moderate increase in nitrogen due to forest activities can be locally beneficial by increasing primary productivity.

Determination of acceptable increases in stream nitrogen concentrations also requires consideration of the downstream uses of water. Streams that flow into oligotrophic lakes and streams used for water supply purposes may require more frequent monitoring and more stringent standards. Continuous discharge measurements are essential for any monitoring that involves calculating total load, while simultaneous discharge measurements are needed to properly interpret any instantaneous measurements of nitrogen concentrations.

2.5.2 PHOSPHORUS

Definition

In natural waters phosphorus can be separated into two fractions, dissolved and particulate (Leonard et al., 1979). Dissolved phosphorus is found almost exclusively in the

form of phosphate ions (PO_4^{3-}), and these bind readily with other chemicals (Hem, 1970). There are three main classes of phosphate compounds—orthophosphates, condensed phosphates, and organically bound phosphates. Each of these can occur in soluble forms (i.e., dissolved phosphorus) or be bound to particulate matter (APHA, 1976). In general, only orthophosphates are readily available for biotic uptake. The common analytic tests are total dissolved phosphate, which includes all three classes of phosphate compounds, and dissolved soluble and plant-available phosphate, which is essentially equivalent to orthophosphate.

Since dissolved phosphorus can be readily transported to the drainage network by surface and subsurface flow, the forms and concentrations of dissolved phosphorus in aquatic systems are directly related to the terrestrial phosphorus cycle. Like nitrogen, most of the phosphorus in terrestrial ecosystems is insoluble and thus immobile. Losses of soluble phosphorus due to leaching in Douglas-fir and silver fir forests represent an insignificant portion of the total phosphorus on the site (Cole, 1979). Annual phosphorus budgets for four forested coastal watersheds range from a net gain of $0.1 \text{ kg ha}^{-1} \text{ yr}^{-1}$ to a net loss of $0.3 \text{ kg ha}^{-1} \text{ yr}^{-1}$ (Feller and Kimmins, 1979). Subalpine watersheds subject to heavy spring snowmelt and periodically intense rainfall may not conserve phosphorus (Leonard et al., 1979).

Any unbound phosphate ions that enter into streams and lakes, or which are released by microbial decomposition, are readily taken up by aquatic plants and microorganisms. The rapid biological uptake and the ease of chemical bonding explain why phosphate concentrations in natural waters generally are very low (Hynes, 1970; Hem, 1970; APHA, 1976). Mean annual phosphorus concentrations in small forest streams on the west slope of the Cascades typically are less than 0.06 mg L^{-1} (Brown et al., 1973; Feller and Kimmins, 1979; Harr and Fredriksen, 1988; Martin and Harr, 1989).

Particulate inorganic phosphorus is mineral in origin and enters the stream channel primarily by soil erosion and sediment transport. Particulate organic phosphorus comes from a variety of sources and can enter the stream channel through fluvial transport or direct deposition. In slow-moving streams and lakes, there may be a net loss of phosphorus through the settling of sediment and organic material (EPA, 1986b).

Phosphorus is not considered to be limiting to forest growth in the Pacific Northwest and is rarely applied as fertilizer (Gessel et al., 1979). Forest soils have a high capacity to bind dissolved phosphorus through a variety of 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.

Relation to Designated Uses

Phosphorus is an essential nutrient for plant growth. However, an increase in plant-available phosphorus may not necessarily increase primary production, as other factors may be limiting (e.g., Scrivener, 1988). In small forest streams light is often the key limiting factor (Gregory et al., 1987). In contrast, larger streams and lakes are light-saturated, and in these aquatic systems nutrients tend to limit primary production.

In aquatic ecosystems phosphorus is usually the limiting nutrient (e.g., Mohaupt, 1986). A general rule of thumb is that the optimal nitrogen to phosphorus ratio for primary production is 16:1; a lower ratio suggests that nitrogen is limiting, while a higher ratio indicates that phosphorus is limiting. A survey of streams in western Washington indicated that phosphorus was more likely to limit primary production in glacial streams and in streams draining granitic areas, while nitrogen was more often limiting in streams draining volcanic landforms (Thut and Haydu, 1971). As discussed in Section 7.2, an increase in primary production normally will stimulate invertebrate and fish production.

The desirability of an increase in primary production depends on the local and downstream uses of water. In many forested areas an increase in stream production might be considered beneficial. Often, however, these streams flow into lakes and rivers that are extensively used for fishing and recreation. Increased primary production in downstream receiving waters due to nutrient enrichment (eutrophication) can impair certain designated uses. Adverse effects can include changes in water chemistry, reductions in dissolved oxygen levels, less recreational use, and a decline in esthetic values.

Naturally occurring phosphate concentrations present few problems with regard to acute or chronic toxicity. Laboratory experiments have shown that fish can accumulate elemental phosphorus from aqueous concentrations as low as $1 \mu\text{g L}^{-1}$, but elemental phosphorus is not present under natural conditions in forested areas (EPA, 1986b).

Response to Management Activities

Studies in the Pacific Northwest indicate that forest management activities are unlikely to substantially increase phosphate concentrations in aquatic ecosystems. Data from small catchment studies in Oregon showed that clearcutting and burning had no effect on phosphate concentrations (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 $<1 \mu\text{g L}^{-1}$ (Scrivener, 1988).

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Other studies have suggested that phosphate losses increase after wildfire, but the increases are relatively small in both relative and absolute terms (Wright, 1981). Widespread application of phosphate fertilizers could increase aqueous phosphate concentrations, but phosphate fertilizers are rarely used in the Pacific Northwest (Gessel et al., 1979).

Human waste can contribute significant amounts of phosphorus to aquatic ecosystems. Dispersed recreation and septic tanks are two common nonpoint sources. Many rural communities are unable to afford the tertiary sewage treatment necessary to remove the primary nutrients and protect water bodies from eutrophication. Livestock and other animals represent another potentially important source of phosphate contamination. The chemistry of phosphorus is such that most of the phosphorus entering into aquatic ecosystems will be in the form of orthophosphates and either sorbed onto soil particles or incorporated into organic compounds. Thus soil erosion can be a primary source of phosphorus, whereas nitrogen usually reaches aquatic systems in dissolved forms (Mohaupt, 1986).

Measurement Concepts

The procedures for analyzing the various forms of phosphorus are discussed in considerable detail in standard references (e.g., APHA, 1989; Stednick, 1991). A comparison of inorganic particulate and dissolved phosphorus can help determine whether the primary source of phosphorus is due to erosion or solute transport. Dissolved phosphorus is the focus of most monitoring programs, and is often separated into soluble and plant-available phosphate (i.e., orthophosphate), and total phosphate. Dissolved organic phosphate can be estimated by subtracting soluble and plant-available phosphate from total phosphate.

Since the analytical techniques are well defined and most forest streams are considered to be well mixed, the primary measurement problems are (1) determining the timing, location and intensity of sampling; and (2) avoiding contamination. Determining the type and intensity of measurements largely depends on the monitoring objectives; this is discussed in detail in Part I, Section 4. Contamination is a serious problem because the natural concentrations of phosphorus in forested streams are $<0.02 \text{ mg L}^{-1}$ (EPA, 1986b; Kunkle et al., 1987).

If the primary monitoring concern is the total load of phosphorus, both particulate and dissolved phosphorus should be sampled. More intensive sampling will be needed during high flows, which is when most of the particulate phosphorus will pass by the monitoring station. Continuous discharge data will be needed to calculate the total load (concentration times discharge). It may be possible to establish a quantitative relationship between particulate phosphorus and turbidity, and this could substantially reduce the costs of monitoring particulate phosphorus.

Analysis of the effects of a particular management activity generally will entail more limited sampling at carefully selected locations. For both project and trend monitoring, an analysis of total phosphate often will suffice. Simultaneous discharge data are needed to help interpret the temporal variability in concentration and optimize sampling.

Standards

A national water quality criteria of $0.10 \text{ } \mu\text{g L}^{-1}$ of elemental phosphorus has been established for marine and estuarine waters. No standard has been set for phosphate concentrations in freshwater because the threat of eutrophication is so location-specific (EPA, 1986b). Some of the specific factors that may affect the desirable phosphate level include the extent to which phosphate is actually limiting primary production, the technological feasibility of phosphate control, and the beneficial uses of the water body.

Although specific standards have not been established, EPA (1986b) has made some general recommendations regarding the maximum concentration of phosphorus in streams and lakes. To prevent eutrophication, total phosphates as phosphorus ($\text{PO}_4^{3-}\text{-P}$) should not exceed 0.025 mg L^{-1} for any lake or reservoir. (Total phosphates as phosphorus refers to the mass of phosphorus atoms per liter; phosphorus atoms represent only 32.6% of the total mass of phosphates per liter.) Where streams enter into reservoirs or lakes, the concentration of total phosphates as phosphorus should not exceed 0.050 mg L^{-1} . Mackenthun (1973) suggested that total phosphorus concentrations should not exceed 0.10 mg L^{-1} for streams that do not flow into reservoirs or lakes. Concentrations of $\text{PO}_4^{3-}\text{-P}$ above 0.10 mg L^{-1} may interfere with the coagulation process in water treatment plants (EPA, 1986b).

Current Uses

Phosphate concentrations are not a regular part of most water quality monitoring programs in forested areas. This omission is due to the very low background levels of phosphate in forest streams in Alaska and the Pacific Northwest, and the relatively minor impact of most forest management activities on phosphorus concentrations. However, phosphorus often is the limiting nutrient in aquatic ecosystems, and this may make it the primary constituent of interest in some water bodies. Currently the load allocation process (Part I, Section 1.4) has been initiated for phosphorus in both the Tualatin (Oregon) and Spokane River basins. The need to quantify all natural and anthropogenic sources means that phosphorus concentrations in streams emanating from forested areas also must be monitored.

Assessment

Phosphate monitoring in forested areas is most likely to be necessary when eutrophication threatens downstream

rivers, lakes, or reservoirs. Although particulate sources may need to be evaluated, it is much easier to monitor total phosphorus only in downstream receiving waters most likely to be affected by eutrophication. Changes in dissolved phosphorus concentration usually are easier to detect in lakes and reservoirs because the concentrations are less variable over time (i.e., it is much less dependent on discharge), and changes in phosphorus concentrations tend to accumulate over time. This accumulation of phosphorus is easier to detect than smaller, incremental increases in phosphorus inputs.

More frequent monitoring of phosphates in forest streams may be necessary if phosphate fertilization becomes a common practice. Under these circumstances the guidelines for monitoring would be similar to the guidelines suggested for monitoring nitrogen inputs from forest fertilization (Section 2.5.1).

Although phosphate monitoring may not be a necessary component for most water quality monitoring programs in forested areas, additional data on phosphate concentrations can be useful. Information on the normal range of values in different ecoregions and the temporal variation within streams will be helpful if monitoring phosphorus loads becomes necessary. Similarly, long-term data on phosphorus inputs, nitrogen inputs, and surrogates for free-floating algal biomass (e.g., chlorophyll *a*) could help determine the limiting factors for plant growth in larger water bodies, and hence the likely response of a particular water body to a projected change in light, nutrients, or other factors.

2.6 HERBICIDES AND PESTICIDES

Definition

Pesticides are chemicals used to control undesired plant or animal species (i.e., pests). Chemicals for controlling unwanted plants are called herbicides. Chemicals for controlling animal pests commonly are called pesticides, even though this is not consistent with the strict definition given above. Other terms, such as insecticides, are specific to particular groups of animals. Since herbicides may have quite different effects on water quality and aquatic ecosystems than the chemicals used for controlling animal pests, there is a need to distinguish between these two broad classes of pest-control chemicals. These *Guidelines* follow the more common and practical approach of using the term pesticide only for those chemicals which are directed against animal pests.

Herbicides and pesticides can be applied directly by ground-based methods or sprayed from aircraft. Since most of the commonly used herbicides and pesticides affect a broad range of organisms, their use has engendered considerable controversy.

Most of this concern has focused on the aerial application of herbicides and pesticides, as it is very difficult to prevent some spray from being applied in or near the stream channels. In addition to the contamination problem posed by overspray and drift, herbicides and pesticides can be transported from the point of application to the aquatic ecosystem by leaching, volatilization, and erosion. In most cases the risk of contamination from these three transport processes is lower than risk of contamination from direct overspraying and drift (EPA, 1977).

The susceptibility of a chemical to leaching depends on its solubility in water and its tendency to adhere to soil particles. Another pathway for these chemicals to reach the aquatic ecosystem is by absorption onto soil particles and subsequent erosion. The dependence of these transport mechanisms on the movement of water means that the potential for contaminating streams and lakes is closely tied to the amount of precipitation, runoff, and erosion following application. High runoff events shortly after application generally pose the greatest risk for loss from the terrestrial environment to the aquatic ecosystem (e.g., Reynolds, 1989). The tendency to use less persistent herbicides (i.e., ones which break down more rapidly) and the relatively low levels of pesticide application suggest that transport from the terrestrial to the aquatic ecosystem is rarely a problem except for a few chemicals such as picloram and atrazine (Fredriksen et al., 1975; NCASI, 1984a).

Relation to Designated Uses

Although herbicide and pesticide contamination can adversely effect several designated uses, the protection of domestic water supplies has the highest priority. For many of the more common herbicides, EPA has recommended a maximum mean concentration for any 24-hr period. These maxima have been derived from combining toxicity tests on aquatic organisms with a safety factor (EPA, 1977). Since most silvicultural chemicals have a half-life of no more than a few weeks, the potential for bioaccumulation is relatively low and the 24-hr standard is adequate. However, herbicides, pesticides, and their intermediate breakdown products are all subject to adsorption by soil or fine sediment particles, and this can affect the relative persistence, toxicity, and biologic uptake of these chemicals.

The possibility of sublethal effects on aquatic organisms cannot be excluded, but the intermittent application, short duration of exposure, and relatively low concentrations all suggest that these effects generally are small or insignificant. A detailed study of the effects of glyphosphate on a stream in coastal British Columbia, for example, confirmed short-term avoidance and stress by coho salmon and two species of invertebrates in an oversprayed tributary; long-term effects were deemed negligible (Reynolds et al., 1989).

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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 thunderstorms 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-

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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. Monitor-

ing 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-

tures 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

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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 $\leq 3-6\%$ 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 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-

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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 absorption 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.

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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 estab-

lished 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 $\pm 25\%$ 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 $\pm 10\%$) (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

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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 con-

cluded 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-

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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.

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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.

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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

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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 ($\pm 10\%$), 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

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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

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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 inter-

gravel 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

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major technique to obtain a bulk sample is to freeze a sample of the bed material using liquid CO₂ 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).

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.,

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.

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

$$ISI = \Sigma D_f / \text{Area.}$$

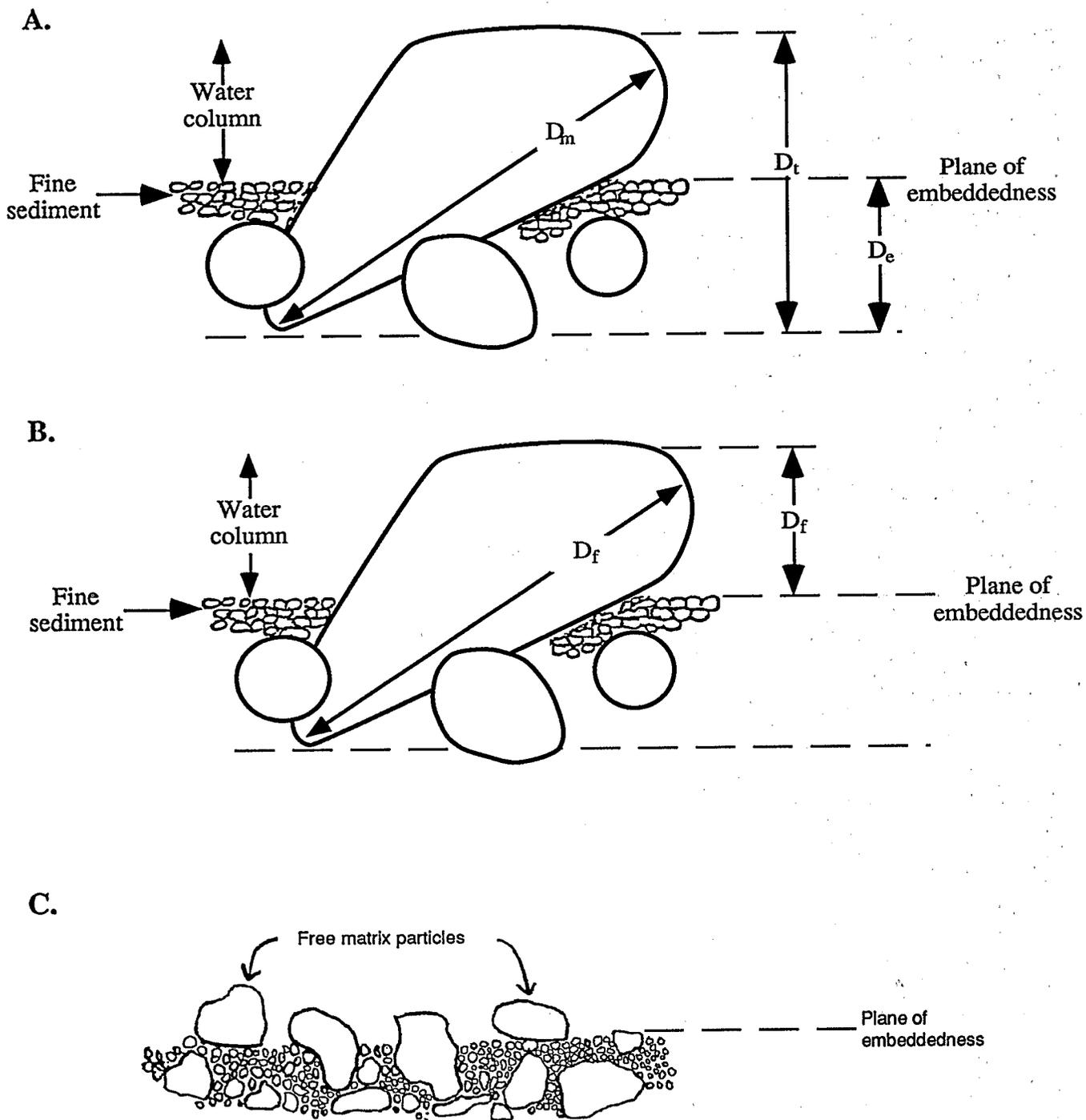


Figure 8. Schematic representation of the three main embeddedness measurements—embeddedness, free space, and free matrix particles. D_m represents the length of the primary axis. A. Embeddedness for a single particle is equal to D_e/D_t . B. Free space for a single particle is equal to D_f (note: $D_f = D_t - D_e$). 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 inter-gravel 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.

Part II

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% confi-

dence 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).

Part II

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 pre-disturbance 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).

Part II

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; McMahan 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 estimates 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

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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

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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

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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

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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.

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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 pre-

sented 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 (Raedke, 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

^aGravel and cobble streambeds are composed predominately of material <25 cm in diameter.

^bWashington 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.

^cOr 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 $44.5 \pm 0.2^\circ\text{C}$. 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 exist (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 ad-

verse 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

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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).

Part II

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 phosphorus, 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).

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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 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 re-

quire 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

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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 eliminate 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 than 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

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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 underes-

timate 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).

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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 biological 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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