A MONITORING STRATEGY FOR APPLICATION TO SALMON-BEARING WATERSHEDS

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The causes of environmental degradation and loss of biodiversity are rooted in society's values and the ethical foundation from which values are pursued (Orr 1992). Solutions are likely to emerge only from a deep-seated will, not from better technology. Adopting biological integrity as a primary management goal provides a workable framework for sustainable resource use, but fostering integrity requires societal commitment well beyond government regulations and piecemeal protection. Such a commitment includes self-imposed limits on human population size and resource consumption, rethinking prevailing views of land stewardship and energy use, and viewing biological conservation as essential rather than as a luxury or nuisance.
—Angermeier and Karr (1994).

"Adaptive management" must be defined as simply "mistakes are inevitable given our rudimentary knowledge of communities and ecosystems, and we will attempt to correct them." Sadly, these "experiments" are increasingly done on landscape scales where any but the smallest of mistakes cannot be corrected, where the cumulative effects of several simultaneous insults must be interpreted, and where political and social resistance prevent timely responses. Perhaps it is time that ecologists projected a more realistic image, i.e., that we don't know enough to truly manage natural resources in any but the simplest ways.
—Schindler (1996).

Terms like "ecosystem resilience," " ecosystem health," "ecological integrity, " and "sustainable development" continue to be bandied about, and are certainly useful as theoretical constructs, but we know of no consistent way of measuring or comparing any of them. When "cumulative effects" are investigated, the interactions of even a few stressors produce horrendously complicated and counterintuitive effects on ecosystems."
—Schindler (1996).

I would emphasize that we have only rudimentary understanding of the dynamics of single populations, and only have hints about the possible behavior of systems.
.....There is abundant evidence of poor or unsuccessful management of ecosystems, but little evidence of successful management (Ehrenfeld 1981, Ludwig et al. 1993, Stanley 1995).
—Ludwig (1996).

There is a critical need to begin multiscaled monitoring—not just monitoring for point-source pollution but monitoring of key features of normal ecosystem function and indicators of the demands imposed by human society.
—O'Neill et al. (1996).

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Part I: Basic Rationale for a Monitoring Plan

What is Monitoring

Monitoring of stream system health is the systematic collection of data on environmental parameters that are linked to beneficial uses and known to be sensitive to land management activities and natural events. Monitoring is used to investigate relationships, validate assumptions and models, assess current condition and follow trends, and improve the basis for management of resources. Such monitoring may involve collecting data within the stream or on the landscape. In-stream monitoring may involve measurement of the physical or chemical characteristics of the channel or water or may be focused on biological responses of fish and other aquatic or riparian-dependent species. Impacts to the stream system occur by direct actions taken in channels or from the indirect effects of activities occurring in riparian areas, floodplains, hillslopes, or on the entire watershed. These effects can be focused at the scale of a single reach or can be more widespread. The spatial distribution of activities and effects requires that monitoring be conducted with respect to this spatial context and also stratified according to representative components of the landscape and stream system. Also, the spatial distribution and temporal characteristics of environmental variables create "habitat" for aquatic biologic communities at a variety of scales that monitoring and data analysis must elucidate. In fact, land management activities vary in type (forestry, range, mining, agriculture, municipal) but their effects on streams can be evaluated in common terms of sediment, water temperature, channel and bank structure, woody debris, pool frequency and depth, biotic community properties, etc.). Effects of land management activities on a stream may be monitored from single sources close to the activity. And because multiple activities are distributed across a watershed in complex overlapping ways and the degradation and recovery responses are often lagged, their effects on the stream and biota may be both cumulative and out of phase with current levels of activities, thereby making it difficult to attribute all current actions with current condition.

Monitoring programs can employ parameters that are slow or fast to respond to perturbations. Variables that respond slowly may be useful to reveal long-term trends but would not permit averting degradation. The most useful monitoring parameters for averting environmental degradation are ones that rapidly reflect the degree of impact to the ecosystem under conditions of gradual environmental change (Schindler 1987). They forewarn changes that, if continued, would result in long-term impairment of biotic health. Physical system monitoring can involve determination of current physical state (e.g., pH of water, streambed fine sediment level) or the rate of change in states (e.g., rate of hillslope evolution or streambank cutting). Indices of change in physical state infer potential for shifts in dependent aquatic species, communities, or ecosystem functions (Schindler 1987). Biotic monitoring parameters can usefully integrate either the existing state of the habitat or recent (relative to life span of the species population or predominant species of a community) trends in habitat condition.

Schindler (1987) noted that ecologists are not very successful in predicting ecological stress from known perturbations or in detecting such stress before the level of environmental degradation is severe and recovery is difficult. Reasons for this limitation include: a focus on maintenance of ecosystem functions (e.g., primary production, community respiration) rather than examining characteristics such as species composition (e.g., loss of predators), loss of sensitive species, invasion of exotics, shift in life table characteristics); use of deterministic models that invariably use inappropriate coefficients or have intractably extensive data requirements; the lack of long-term monitoring data that can be used to establish natural degrees of variation in community characteristics; and a lack of data on biotic systems that can be used to establish baseline targets for unpolluted conditions. Wallace et al. (1996) was successful, however, in employing two biotic indices for stream macroinvertebrates (the EPT or Ephemeroptera + Plecoptera + Trichoptera index and the North Carolina Biotic Index) to indicate change in important ecosystem processes (leaf litter processing rates, organic matter storage, fine particulate organic matter generation and export, and secondary production) in response to chemical pollution. It is unclear how these indices would respond to multiple sources of degradation.

Because biotic performances are a reflection of existing habitat condition or recent trends in condition, and biotic or environmental monitoring may not be able to provide sufficiently "early warning" to ensure that further degradation or irreversible trends do not occur, the most effective means to avoid undesirable biotic trends is to (1) avoid those activities that are routinely linked with these responses (see Rhodes et al. 1994), (2) monitor implementation of activities that pass the screening process (i.e., those considered to have negligible potential effects) to assure they are carried out as planned, and also (3) to use condition and trend in the habitat system to make adjustments necessary to protect or restore habitat conditions. For instance, the level and trends in fine sediment in spawning gravel is a good indicator of survival to emergence (STE) of salmon and trout. Some research indicates that increasing the level of fine sediment in spawning gravels from 0 to 20% fines may result in little decrease in fish survival to emergence but a major decline when fine sediment increases from 20 to 40% (USFS 1983, Bjornn and Reiser 1991). Other research shows a more uniform decline in STE with increasing fine sediment (Chapman and McLeod 1987, see their p. 206). Whether or not there is actually such a marked response threshold, the conclusion remains that increasing levels of fine sediment over natural background levels should be avoided, and if increases between 0 and 20% could actually be detected, they must be used as an early warning index of expected reduction in survival. Recognizing negative trends in habitat condition and avoiding the gradual approach to critical thresholds is important because many effects, such as high levels of fine sediment in spawning gravel or pools, once established, are very slowly reversible and propagate harmful biotic effects for generations in a given species.

In-stream monitoring historically emphasized water chemistry as the key index to stream system health (Karr 1994, 1995). Pure water was considered to provide the conditions necessary for biotic health. However, subsequent aquatic studies revealed that clean water, low in certain nutrients or certain toxics, is not sufficient to ensure diverse aquatic communities. Features such as channel morphology, channel unit distribution, presence of woody debris, abundance and diversity of detrital sources, substrate composition, water temperature and flow regimes, and other factors also are influential in determining biotic health. Because biota are a key beneficial use dependent on water quality and the entire habitat system and because the complex interactions of habitat parameters make it difficult to predict biotic responses, there is good reason to place primary emphasis on biotic monitoring.

The position taken in this report is that pristine streams provide our best reflection of desirable biotic communities. However, even pristine streams may not support historic biotic diversity for several reasons. There may be too few pristine examples of a particular stream type to adequately represent long-term behavior of such systems from a point-in-time sample; fire prevention can disrupt natural processes in pristine landscapes; global effects of ozone depletion, warming, or past hydrological events can eliminate certain species or alter stream conditions for significant time periods; fragmentation of pristine stream systems can weaken their ability to support wide-ranging species. Lacking suitable pristine streams as a robust representation of biotic diversity, one could accept the best of what remains as the standard or allow time for full habitat recovery to take place and then assess biotic composition. It may be possible to establish general expectations for various biotic indices on the basis of river continuum theory (increasing drainage area and related biotic community shifts) for a given ecoregion, but for purposes of providing direct feedback to land managers, we consider it to be of central importance to monitor key environmental parameters such as sediment and water temperature. The chain of causation is typically land use action, local effect on the land, local effect on water quality and instream habitat, effect on water quality and instream habitat along a river continuum, and downstream effect to water quality and instream habitat at a particular site. Effects on biota could occur immediately in toxic spills but may also be progressive as with changes in sediment or temperature regime under impacts to slope stability or riparian cover, or lagged as with downstream sediment effects or the LWD-riparian tree size class relationship. When it is known that low levels of anthropogenic sediment delivery or thermal loading are desirable, it seems logical to us to emphasize first a rigorous screening of land use projects and then a monitoring feedback that is very responsive to trends in, for example, sediment and water temperature, the key immediate outputs from these actions. Monitoring the secondary effects of sediment or temperature on biota provide important additional information, but it may not lead to improved land use as readily. If the intent of land managers is to do their best to provide suitable conditions for coldwater biota, they would first conduct no activities known to increase thermal loading and then would monitor temperature trends. If a temperature increase of 1°F is detected after an activity, there is reason to suspect that the action should be modified and the trend reversed. It is unlikely that biotic monitoring would be sensitive enough to detect the temperature trend until larger land use actions are taken that would take a long time to reverse.

It is also important to monitor parameters that may not be designated as early warning, especially in streams that have suffered from years of degradation. For example, data on current availability of large pools in streams across the Columbia River basin (McIntosh et al. 1994, USFS PNW, unpublished data) attests to a minimum magnitude and type of stream degradation over a 50-year period outside of wilderness areas caused by cumulative effects of activities. These "late warning" monitoring data provide information on a fundamental habitat attribute that is invaluable in long-term trend assessment and feedback to management. It sets the minimum target for recovery and provides an index to a major structural component of the habitat system with slow response time.

Monitoring, as described in this document, is a necessary tool for providing data critical to adaptive management. Such feedback is provided by a combination of determining effectiveness of land management practices, validating predictive models, and evaluating status and trends of key in-channel habitat variables, watershed condition, and the dependent salmon resource. It provides the means to assess whether (1) land management standards are being met, (2) in-channel physical or chemical standards are being met or may be violated in the future given certain management trends, (3) biotic standards are being met, (4) overall in-channel habitat recovery is occurring, (5) restoration and protection of streams and watersheds is adequate for recovery, (6) watershed conditions are improving the habitat for listed stocks, (6) the goals of the Clean Water Act are being met (i.e., to provide fishable, swimmable waters and to protect and restore biotic integrity), and (7) the ESA-listed species are recovering.

This monitoring document is tiered to the Coarse Screening Process (CSP) (Rhodes et al. 1994) for determining the consistency of land use activities with the ESA goals of protecting critical habitat and restoring it where degraded, based on the biological requirements of salmon and existing habitat conditions. Under the guidance of the CSP, minimum acceptable habitat standards are those conditions that lead to maximum survival. Where current conditions exceed these biologically-based standards, habitat condition should not be degraded by management; where habitat condition is not meeting biologically-based standards for any reason (natural or management-related), it should not be allowed to become even more degraded, but moreover, land management actions linked to the degradation should be deferred (not merely tinkered with in some optimization scheme that presumes refined understanding of all ecosystem processes and response rates under management stress).

Although this monitoring plan provides the necessary input to the CSP, it also provides input necessary for effective management of land and aquatic systems at various watershed scales and for tracking the effectiveness of efforts to protect and improve habitat conditions and survival under the ESA. This monitoring plan is not meant to be an exhaustive evaluation of monitoring protocols and parameters; many useful reviews of monitoring variables are available and cited here. This document also does not suggest that land managers can rely on monitoring solely the basic variables selected; monitoring of other parameters may be necessary. It does, however, provide: a framework for monitoring habitat for ESA-listed salmon species at various spatial scales and levels of intensity; a framework for linking population and community data for fish and other biota to habitat on several hierarchical spatial scales; a guide to literature for monitoring and analysis of key habitat factors; a critical analysis of selected habitat factors; and a review of important concepts in aquatic ecology relevant to monitoring.

Monitoring's Role in Managing Resources:Agency and Legal Perspectives

With the possible exception of riparian habitat management, no other subject has had a higher ratio of what has been written about monitoring to what has been accomplished. The history of monitoring land management actions and their effects upon aquatic resources in the Snake River Subbasin is characterized by more rhetoric than implementation[(1)]. The most significant factor

(1) Forest plans for Columbia River basin national forests promised to implement monitoring plans and adjust management actions according to in-stream conditions but have largely failed to commit to this approach. FEMAT (i.e., USFS et al. 1993, USFS and BLM 1994) and PACFISH (i.e., USFS and USBLM 1995) are management

limiting monitoring efforts is ***commitment***. Without it, there is no funding to support and sustain monitoring activities, because there is considerable discretion to devote resources toward monitoring in forest budgets[(2)]. Because monitoring can lead to accountability and changes in output, commodity-oriented management has often relegated the activity to low priority in the budgetary process. Consequently, monitoring in the Snake River Subbasin has had an uncertain, weak, and inconsistent history (see Henjum et al. 1994). Despite the rhetoric in land use plans and laws (NFMA), few management units have long-term monitoring programs and databases. Management leadership has not placed a high priority on getting answers to critical management questions that monitoring can provide. Instead, it appears content to justify status quo management because no data exist that indicate problems, while professing interest in applying science to answering uncertainties. The combination of waiting for a 5-10-year data collection (probably a minimum time frame needed to observe change, given the inherently slow response of many of the variables that federal agencies propose to monitor for critical habitat under ESA; see USFS 1994) to confirm a trend and also unwillingness to adequately fund monitoring programs, ensures a lack of management feedback. Insufficiency of monitoring programs is compounded by (1) a lack of intensity of effort on local sites where activities occur, (2) an unwillingness to change management before major damage is proven at high levels of statistical significance, (3) a lack of broad geographical coverage in monitoring and synthesis in reporting, (4) an unwillingness to extrapolate experience from one site to other similar sites, (5) a tendency to monitor only variables known to be slow to respond to environmental change, and (6) a tendency to permit actions known to negatively impact the salmon habitat system to occur while at the same time arguing that these effects will be difficult to distinguish from long-term ranges of performance of "unperturbed" systems.

The cavalier treatment of monitoring and the related absence of commitment to achieving specific habitat conditions appear to still be in evidence in federal land management plans, but the urgent need for accurate assessments of habitat condition before proceeding with potentially damaging actions highlights the need for a swift end to this attitude. Even though the USFS, BLM, USFWS, SCS, ODFW, and WDG all signed an agreement in 1979 to conduct land management to achieve no more than 15% fine sediment in stream substrate (see Appendix A), this commitment has been ignored and is no longer part of forest management as a standard, even in habitat of listed

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strategies intended to improve existing land resource management plans in order to reduce degradation of salmon habitat and to meet requirements under the Endangered Species Act. FEMAT tiers to a monitoring program but does not commit requiring specific standards. Both FEMAT and PACFISH proffer habitat "objectives" instead of standards, and do <u>not</u> commit to adjusting management based on monitoring results to meet objectives (Rhodes 1995). FEMAT recommends primary emphasis on implementation monitoring. PACFISH explicitly states that effectiveness monitoring for habitat objectives will not occur. Neither FEMAT nor PACFISH require that land use actions proceed only when monitoring data are available or if monitoring accompanies the action. Both Rhodes et al. (1994) and Henjum et al. (1994) recommend that management activities should be contingent on monitoring data that indicate that habitat conditions are not degraded.

(2) The lack of budgeting for monitoring, though, is not solely the fault of individual forests; it is a regional- and national-level example of neglect. PACFISH does not consider standards to be hard targets, so the incentive to be serious about monitoring is absent. Also, Congress allocates funds for massive road construction on forest land, but the environmental costs of the road building legacy are not supported, nor are the costs of monitoring the environmental consequences.

species (see USFS et al. 1993, USFS and USBLM 1994, USFS and USBLM 1995, i.e., FEMAT and PACFISH). The listing of the Snake River salmon under the Endangered Species Act (ESA), the potential ESA listing of steelhead and bull trout, litigation under the Clean Water Act (Idaho Sportsmen's Coalition v. Carol Browner et al.), PACFISH, ICBEMP (Interior Columbia Basin Ecosystem Management Plan), and increased public awareness have pressured federal and state agencies to make stronger commitments towards monitoring. It is within this evolving context that the monitoring strategy for Snake River salmon is presented.

Monitoring alone, however, will not ensure that land management will be effective in restoring degraded ecosystems and meeting the biological requirements of ESA listed species. Although monitoring can provide data that are needed for informed decision-making, land management is unlikely to improve unless: (a) appropriate data-driven goals are set and, (b) management adheres to these goals and changes as needed based on monitoring feedback. Even under these ideal conditions, monitoring data typically provides information leading to adaptive management only after some level of degradation has occurred. For management to be effective, it should make use of existing knowledge concerning linkages between land management actions and habitat alterations so that habitat damage can be prevented before it occurs. Monitoring that indicates no or little recovery during benign climatic periods may forewarn significant reversals of condition during future adverse climatic conditions given a continuation of management practices. Actions that have a likelihood of causing adverse impacts can be screened out before they are implemented or allowed to continue. Assumptions about actions that are deemed to promote or not impede habitat recovery must be validated by monitoring. This is part of the purpose of the CSP and should be at the heart of any effective approach to protecting salmon and their habitats under the ESA. Monitoring data should be oriented to providing new understanding of the recovery rates achievable under consistently applied management procedures. Further, it is essential that biologically-based habitat standards be adopted that provide full beneficial use protection targets for habitat restoration. Such standards will also provide clear decision criteria for land management. Adoption of targets that do not provide excellent desired conditions for degraded habitats will obviously result in land being managed for degraded habitat conditions. It may appear reasonable to manage for habitat conditions based on *a priori* assumptions of attainability, stream reach by stream reach. However, as elaborated in Rhodes et al. (1994), examples of high quality habitat in larger salmon-bearing reaches of the Snake River or elsewhere in the Columbia River system are virtually non-existent. Attainability is then a matter of pure supposition. A management prerogative to set low, easily achievable targets (e.g., because they are the lowest common denominator for a region) is not likely to protect ESA-listed salmon species, even though they may appear to fall within the wide bounds of conditions in "least perturbed" streams. Long-term application of the best possible management practices (emphasizing relaxation of anthropogenic stresses) aiming for, at a minimum, biologically based standards that reduce salmon mortality to near-natural background levels, may begin to reveal attainable conditions.

In the ESA Section 7 consultation process, the National Marine Fisheries Service has committed to reviewing actions singly and in combination to assess whether each action will result in some reduction in salmon mortality relative to a 1986-1990 base period. The CSP (Rhodes et al. 1994) is offered as a means of performing this screening of actions. The monitoring plan provides information needed for adaptive management in the event that proposed land management standards of the CSP are not stringent enough to maintain or initiate improving trends. It is also a means of determining whether biologically-based standards are being met in the stream or more broadly across all the streams upon which the listed species is dependent. The monitoring plan also addresses conditions in non-salmon-bearing tributaries to salmon-bearing reaches as a means to arrest degradation and its causes before habitat damage can propagate downstream. While a portion of the monitoring plan is specifically linked to the CSP, the entire monitoring approach is aimed at assessing attainment of ESA goals, including prevention of adverse modification to critical habitat and degree of improvement in habitat and population survival prior to delisting.

Ecological Foundations for Creation of Monitoring Framework: Habitat/Species Linkages

A well designed monitoring plan for salmon habitat and salmon response must track the condition of vital habitat attributes and key parameters that control and express the long- and short-term behavior of the habitat system (e.g., on- and off-site impacts of activities and conditions over time, change in habitat quality, and change in fish response). The linkages coupling components of a watershed, including the land-aquatic linkages as expressed in Rhodes et al. (1994) (see Appendix B) illustrate that salmon habitat includes entire watersheds and systems of watersheds. Processes operating on watershed hillslopes generate water and sediment input regimes that shape stream channels. It is not possible to disassociate the stream from its watershed. Salmon habitat depends on the combined functions of all the watershed's components such as hillslopes, floodplain, and riparian zone. In other words, salmon habitat is not just systems of riffles and pools but includes the entire set of watershed processes that produced and maintain this system. When numerous watershed processes are altered, these complex spatially varied interactions are reflected by an altered set of in-channel conditions that may not provide high quality salmon habitat. In-channel habitat conditions expressed in by Rhodes et al. (1994) as essential and auxiliary coarse screening variables (see Appendix C) and the linkages of these conditions to perturbations in the watershed (see Appendix B) constitute the starting point for creating a spatially structured in-channel and out-of-channel monitoring program and postulating cause-effect relationships.

Because salmon are anadromous species and migrate within a basin during their freshwater development, monitoring the condition of their habitat and attendant survival response involves assessing: the quantity/quality of habitat units supporting each life history stage; the connectedness of these units; the degree of temporal variation in habitat parameters (e.g., water temperature fluctuation, alteration of seasonal means, accumulated degree days, alteration of critical high and low values); and the quantity/quality of all sets of similar habitat units within the current or historic range upon which the species currently or potentially depends. The latter, spatial aspect of habitat monitoring involves whether the species has numerous, well-dispersed habitat units supporting each life stage that are of high quality and sufficient quantity to prevent creation of population bottlenecks.

The linkages among components of the landscape and the stream system plus the ecological value of various instream habitat types to particular salmon lifestages suggest a logical spatial framework for monitoring fish populations and communities. A coarse scale framework for monitoring salmon habitat is given by a subdivision of the Columbia River Basin into watersheds of decreasing size (Table 1). We recommend that, at a minimum, habitat conditions in all 4th and 5th order watersheds (approximately secondary and primary tributaries) occupied (currently or historically) by the listed species be monitored as "logical production units." Significant salmon habitat occurs at this spatial scale. In addition, watersheds of this spatial scale constitute, what may be for the interim, the upper size limits for which land managers can apply certain technical management tools. For example, the R1/R4 sediment model and variants are used in the Idaho batholith to estimate management-induced sediment delivery; data from this scale can be used to make needed improvements. Aggregation of available habitat and watershed data at a greater geographic scale (e.g., ≥6th order or approximately large watershed scale, see Table 1) may be useful for some purposes. Land management actions in 0- to 3rd order watersheds that are directly tributary to ≥6th order reaches should also be screened for compliance with land management standards relative to their potential effects on these larger order reaches. Monitoring in these low order tributaries is important for providing earliest warning of adverse conditions propagating downstream into salmon habitat and to provide the greatest resolution of connections between land use and channel effects.

Although larger order stream reaches should benefit over the long-term from improved management in smaller order tributaries (e.g., application of CSP screening recommendations designed to avoid damage and allow habitat recovery provide excellent habitat conditions in salmon-bearing reaches), the ability to link habitat conditions with individual and combined upstream actions becomes increasingly difficult with expanding geographic scale. In addition, many habitat parameters may be adequately protected by applying land use standards coupled with monitoring of spawning and rearing conditions in secondary and primary tributaries with attendant management adjustments needed to improve habitat (e.g., reducing sediment loads where fines are greater than 20%). It is assumed that rigorous screening of planned activities and adaptive management based on monitoring at the 4th to 5th order scale will lead to improvement in in-channel conditions from downstream reaches to their headwaters. Improved conditions at the 4th to 5th order scale should be translated downstream but the potential for degradation of ≥6th order reaches by local activities or upstream cumulative effects necessitates monitoring large rivers and their contributing watersheds (i.e., monitoring other than that needed for the CSP). It is often assumed that adequate habitat conditions in spring chinook habitat will confer needed conditions to downstream fall chinook habitat [e.g., see the National Marine Fisheries Service's (NMFS 1996) draft biological opinion on salvage logging in the South Fork Salmon River, Idaho] but this may not always be the case.

The primary purpose of the Coarse Screening Process (CSP or Rhodes et al. 1994) is to determine whether an activity is consistent with protection and improvement in habitat and salmon population survival within a watershed, given existing habitat conditions and trends. The ESA requires such a determination under Section 7. Monitoring data in salmon habitat of a primary tributary might indicate that in-channel habitat standards are not being met. The cause for this situation is often largely the result of cumulative effects from all contributing secondary tributaries. This would, according to the CSP, require application of more stringent land use standards to restore conditions in the primary tributary. The watershed of the primary tributary (e.g., 5th order reach) has secondary tributaries (i.e., two 4th order watersheds). If a secondary tributary meets habitat standards, no actions should be taken that would degrade these conditions. The standards might be met because the secondary tributary watershed is roadless. If so, it should remain roadless until it is documented that recovery has occurred in >90% of watersheds in the entire Snake River basin, according to CSP recommendations. If the secondary tributary does not meet in-channel conditions, land management standards should be applied that will result in rapid recovery of the secondary tributary reach as well as the entire contributing watershed. Implementation of appropriate land management standards throughout the primary tributary watershed is expected to lead to recovery of in-channel conditions in all salmon-bearing reaches.

Table 1. Consideration of scale in the monitoring framework.

Scale	Monitoring Emphasis	Purpose	Example Watershed
region	continuity, commitment, biodiversity, species distribution, data acquisition and analysis	regional status report on aquatic health, aquatic production, and endangered species protection	Pacific Northwest
basin subbasin	flows, fish survival, fish passage, temperature, adult escapement, reservoir sediment deposition and transport	fall chinook population and habitat trends	Columbia River Snake River
large watershed	flows, temperature, substrate sediment, adult escapement	fall chinook population and habitat trends	Clearwater River
primary tributary	flows, temperature, substrate sediment, pools, juvenile abundance, adult escapement, channel morphology, land use, channel types,	spring chinook trends, cumulative effects, selected research (e.g., overwinter habitat use, migration timing)	Lochsa River
secondary tributary	flows, sediment yield and delivery, temperature, substrate sediment, LWD, sediment storage, pools, cover, riparian vegetation, juvenile abundance, adult escapement,	spring chinook and steelhead trends; trends in habitat condition; trends in salmon survival; habitat-fish response linkages; model validation; aggregate effects of land management actions	Fish Creek, White Sand Creek, Brushy Fork, Crooked Fork
headwater tributary	flows, sediment yield and delivery, temperature, substrate sedimen,t LWD, sediment storage, pools, cover, riparian vegetation	model validation, effects of individual projects(causative factor research)	Mex Creek (tributary to Fish Creek)

Part II: Outline of a Monitoring Plan and Its Parameters

The primary objectives in the proposed monitoring plan are to document compliance with NMFS' ESA mandates and policies, to provide necessary data upon which rational decisions can be made regarding land management, to document recovery trends in habitat and salmon populations, and to improve knowledge on linkages among land use, salmon habitat and salmon survival and distribution. Because this monitoring plan is intended to complement the CSP (see p. 4), a major focus is on the CSP parameters. However, it can provide a template for collecting the data needed under any rational framework for tracking the effectiveness of salmon habitat protection under the ESA. The CSP is directed at analysis of habitat conditions in salmon-bearing watersheds. The CSP protocol has a focus on habitat monitoring of stream reaches supporting salmon but requires reconnaissance at the watershed scale and inventory of the entire riparian system and stream channel network. Monitoring of salmon-bearing watersheds involves assessment of condition on 4th to 5th order watersheds (approximately secondary to primary tributaries and up to the level of subbasin for certain types of monitoring and analysis. For most purposes the scale of secondary to primary tributary constitutes a logical salmon production unit, but current or historic habitat often includes larger order reaches (and sometimes smaller ones). In these cases, in-channel habitat standards should be extended to these other reaches. Analysis at a scale of $\geq$6th order will involve aggregation of data collected on all watersheds of lower order within it.

In addition to monitoring of salmon-bearing watersheds to implement the CSP, land management cannot effectively meet in-channel habitat standards in salmon-bearing reaches nor avert habitat degradation without monitoring and feedback occurring in non-salmon-bearing headwater tributary reaches. Also, other types of monitoring are essential, such as to evaluate physical or biological trends in specific reaches (salmon- or non-salmon-bearing), to validate important assumptions and models, such as those used in the CSP and other management models, to establish habitat baseline and trend for watersheds and entire stream systems, and to investigate stress/response under defined conditions (i.e., monitoring effects of activities on hillslopes at the site of an action or in the channel). The framework of the monitoring plan represents what we take to be a spatially sufficient strategy for addressing a wide-range of ESA concerns.

We propose a monitoring framework composed of three levels of data collection and analysis to address differing information needs in a complementary fashion with the appropriate level of resolution. Within each level, monitoring is directed at salmon- and non-salmon-bearing streams and their watersheds. For any of the three levels, monitoring of salmon-bearing watersheds emphasizes the integrated nature of a watershed and its control on salmon habitat quantity and quality. However, in the effort to proactively protect salmon habitat, an equal monitoring effort needs to be directed at non-salmon bearing watersheds that are tributary to salmon-bearing reaches. The theme for the three levels of monitoring in these non-salmon-bearing headwater streams is to concentrate monitoring effort close to the action—i.e., where the majority of impacts are being generated and where environmental signals are clearest and most immediate. This direction satisfies two important requirements: 1) it provides an ***"early warning system"*** in that impacts are measured near the sources and not after the fact in salmon habitat, and 2) the dilution effect, increased complexity, and overlapping patterns encountered with large systems (multiple effects and variables) are avoided. Monitoring within an ESA context strongly predicates an "early warning system." Far too many monitoring programs have concentrated their efforts in large systems and after habitat had been extensively degraded with limited opportunity for rapid recovery (e.g., McIntosh et al. 1994).

In the literature monitoring has been defined as consisting of various types of data collection and analysis such as inventory and assessment, baseline, trend, implementation, compliance, effectiveness, project, and validation monitoring. Definitions provided by MacDonald et al. (1991) for each of these types of monitoring are helpful. These authors provide a flowchart that links the various monitoring types in an adaptive management feedback loop system centered around application and refinement of BMPs, protection of beneficial uses, and meeting water quality standards. As they usefully point out, there should not be many separate kinds of data collection going on independently. Information collected in project monitoring, for example, should be applicable for other objectives as well if the overall monitoring program is well-designed. Despite the importance of this message, the USFS and other land managers tend to relegate implementation and effectiveness monitoring to the national forest staff and validation monitoring to range and forest experiment station research staff. Trends are not tracked for long periods because each data collection effort is considered to be different (i.e., using different methods, frequencies or spatial distribution of sampling, etc.). Data are seldom synthesized or presented graphically on a forest-wide basis showing compliance with standards. Annual forest monitoring reports from Columbia River basin national forests are typically devoid of synthesis or representation of trends. This lessens overall accountability. The current policy of the USFS in FEMAT and ICBEMP is to emphasize qualitative habitat management objectives (soft targets) and rely on a vague intent to maintain watershed processes within a natural range of variation. New and existing logging, mining, and grazing are allowed under FEMAT without regard for current habitat conditions (Rhodes 1995). Without firm, numeric habitat standards, and management linked to monitoring and attainment of these standards, there can be no adaptive management aimed at providing high quality fish habitat.

This monitoring framework (see Part III, beginning on p. 22) attempts to diverge from the conventional federal system of totally separating implementation, effectiveness, and validation monitoring. In development of this framework, it was considered that **inventory**, **assessment**, and **baseline** monitoring were all synonymous. Inventory and assessment might be considered as either exploratory monitoring, such as when **pilot** monitoring is done prior to adoption of a full monitoring program, or as a single purpose program of short duration. However, for simplicity, it is more reasonable to consider it as an initial evaluation of habitat condition, that when continued, will constitute **trend** monitoring. Evaluation of in-channel habitat, riparian, hillslope, or watershed condition trends over short or long time periods can provide indications of **effectiveness** of management practices (either single practices carried out in single **projects** or multiple activities occurring at different times throughout a watershed). In-channel habitat condition evaluated at a point in time or as trends can provide information on compliance of land management activities or entire programs with water or habitat quality standards.

MacDonald et al. (1991) suggest that the data set used to develop a management model should be different from that used to **validate** the model. Although this caveat is important, it is probably not frequently violated in practice because most land management models are not calibrated to every individual stream. Consequently, for the most part, trend data can probably become part of a model validation. This will require collection of more than simply the predicted variable in the model and could require more frequent, precise, or more highly spatially distributed data collection. Even so, coordination of data collection efforts and cross-calibration of methods would eliminate duplication of effort and allow more data from one level of monitoring to be used at other levels.

Implementation monitoring is one form of monitoring that does not fit onto the continuum from baseline to effectiveness and trend to validation monitoring. It is important to determine whether certain management practices were employed simply to document what the management action was and when it was employed. This is little more than recording the conditions of a management (or experimental) treatment. This step is vital in order to adequately interpret the subsequent habitat condition via effectiveness or validation monitoring.

In development of the proposed monitoring framework, we chose to recommend three levels of monitoring. Even though distinctions among **implementation**, **effectiveness**, and **validation** monitoring appear to be relatively clear, it did not seem to be most useful and efficient to identify the three levels with these types of monitoring, because there are many means of **analyzing** or **synthesizing** environmental data that must be effectively distinguished by levels. For example, an **extensive** data collection may cover a large watershed area or stream length and be conducted over a short or long time period (involving numerous repeat measurements). **Intensive** data collection may be narrowly focused on a portion of a watershed or stream (e.g., a reach, channel type, channel unit) and can vary in the time period over which measurements are taken (see Table 2). The extensive-intensive dichotomy can be considered to be a distinction between a **synoptic** versus a **localized** view. The synoptic view can be comprehensive in spatial coverage but also can integrate **numerous interacting variables**. The localized view can take a portion of the system and focus on a more specialized set of variables for a special purpose. An example might be measuring the effect of vegetation types in modifying air temperatures at ground level in a riparian zone as opposed to monitoring hourly regional climate for a year using major interacting variables (precipitation, solar radiation, temperature, etc.). This may be merely a relative distinction based upon an arbitrary scale, such as that created by our monitoring framework. The distinction between synoptic and local coverage can also be made on the basis of the **number of variables** measured. Our framework is based upon fewer variables at Level 1 and more at successively higher levels. More variables may be needed at Level 3 than at Level 1 because Level 3 emphasizes **correlation** among methods to measure the same parameter (e.g., fine sediment). In addition, at Level 3 there is more need to measure associated variables to detect interaction or correlations in response (e.g., to measure fine sediment deposition concurrent with channel widening and pool loss); or to measure the state of variables that may control the expression of a particular response (e.g., the influence of percentage soil moisture saturation on surface erosion into streams rather than merely erosion trends alone).

From Level 1 to Level 3 monitoring can incorporate an increasing **precision** in data collection, although precision varies greatly depending upon the variable measured. For example, if

Level 1 employs visual estimates of surface fine sediment and Level 3 employs a combination of **ocular** surface fine sediment estimation and dry sieving of samples collected by core or shovel to the depth of egg deposition, the Level 3 sampling should provide cross-correlation of methods and also the more precise and accurate estimates for a site. If a given

Table 2. Brief summary of characteristics of three levels of monitoring described in this monitoring framework.

Characteristics	Level 1	Level 2	Level 3
Primary objective	Determine broad scale condition and trends in core habitat attributes and cumulative effects and in salmon populations.	Determine local condition and trends of habitat and fish populations in geographically distributed representative or strategic sites.	Validate significant habitat management models and population response models; determine stress/response relationships; process investigation (mechanisms, sources, cause-effect), cross-correlation of methods
Scale	1:100 to 1:250K	1:25 to 1:50K	1:10 to 1:50K
Relative geographic coverage	Extensive	Intensive	Extensive or intensive
Scope of analysis	Synoptic; multiple response of sensitive variables along a river continuum to cumulative actions; aggregation of data from finer geographic scales or from more detailed analysis (i.e., Level 2 or 3).	Correlative, detailed, local; impact analysis on a project site and in the immediate channel area.	Mechanistic, detailed, local.
Geographic coverage	All salmon-bearing watersheds; all component tributary watersheds; all spawning reaches; all rearing reaches; total stream network condition.	Every *j*th tributary. Every *n*th reach of each channel type; every *k*th channel unit. Or representative, strategic, sensitive, or stable reaches for a given tributary.	Selected watersheds (salmon-bearing, non-salmon bearing) of a given type. Selected hillslopes, floodplains, riparian zones, channel type. Project sites.

Characteristics	Level 1	Level 2	Level 3
Number of variables per issue of concern	Few	Many; often requires multiple methods per variable to ensure data reliability	Many; often requires measurement of associated dependent variables or controlling variables.
Nature of methods used	Rapid	Labor-intensive	Labor-intensive
Frequency of sampling	Every 5 years for watershed and riparian zone variables. Every year for in-channel sediment; continuous for water temperature; annual for redd counts; every 5 years for juvenile counts.	Every year to every two years.	Continual to annual on fixed reference sites. In relation to annual or extreme event streamflow levels. As required by specific monitoring issue.
Approach to implementation monitoring	Evaluate application of CSP land management recommendations; land allocations for major land units (e.g., designation of riparian and roadless reserves) and planned management actions for these units.	Evaluate site-specific application of land management practices	N/A
Approach to effectiveness monitoring	Monitor habitat and salmon response to entire management program or application of recommendations for land management under the ESA (e.g., the CSP) throughout the watershed and stream system	Monitor habitat and biotic response to individual or combined actions in a project on site or locally off site.	N/A
Approach to sampling	Measurement of the habitat or biotic variable for the entire area of the logical salmon production unit (watershed).	Sub-sampling within reaches based upon hierarchical stratification of land/water units; identification of representative reaches or sites for documentation of effects to entire system	Total measure or sub-sampling

Characteristics	Level 1	Level 2	Level 3
Interrelation of levels	Monitoring conducted specifically at Level 1 using coarse-grained methods (rapid method; remote sensing, etc.) sets geographic framework for Level 2 and 3 monitoring; identification of representative reaches or sites; mapping of current status or distribution of problems which aid in developing restoration priorities.	Provides ground-truth for coarse-grained monitoring in Level 1. Aggregate and synthesize the intensive analysis of Level 2 or 3 to provide a Level 1 monitoring overview if data are available. Generates hypotheses for analysis at Level 3.	Validation of methods used in Level 1 and 2 monitoring via intensive monitoring or application of more technologically advanced coarse-grained methods makes it feasible to apply more rapid and effective methods at Levels 1 and 2.
Utility of monitoring in an ESA context	Derive basic data needs used in the CSP to assess trends in habitat variables influencing salmon survival; habitat condition can be used to assess whether proposed land management projects are consistent with the need to protect and restore listed species habitat and populations.	Data collection needed to get refined estimate of whether recovery is beginning or degradation is occurring; spatial distribution of the recovery process.	Data collection to improve land management standards and in-channel standards; to refine understanding of fish population/habitat relationships; rates of recovery; improve BMPs.

method is employed at two different monitoring levels (e.g., Level 2 and 3), one would expect a higher precision at the higher level because of a greater number of samples. The **accuracy** of estimates for the effort expended, however, remains to be determined in monitoring programs. For example, in Level 1 monitoring, to get a total estimate of riparian cover for an entire riparian system, **remote sensing** data (aerial photo, satellite imagery, thermographic analysis) may provide the best means to achieve high accuracy for the time spent. Attempting to get an estimate of average riparian cover for a large geographic area employing only **ground-based** measures with statistical sampling procedures may be less accurate for a certain level of effort because of spatial variation. Ground-based estimates of riparian cover in well-defined stream reaches made in Level 2 or 3 monitoring may be useful as part of the ground-truthing of the remote sensing estimates of Level 1.

Effectiveness of a particular action can be assessed by evidence that in-channel habitat, riparian, or hillslope conditions improve or continue to meet standards during and after the action. It can also be evaluated in terms of whether inputs to or outputs from a particular area change significantly after the action. For example, sediment yield from a project site on a hillslope may increase relative to other similar, unaffected sites. Also, sediment yield to a stream channel may increase after this action or a group of sediment-producing actions in the watershed. This type of monitoring provides the most direct reflection of potential future habitat trends. Habitat trends in the channel close to the source of these activities is the next best indicator of effectiveness of an action or group of related sediment-producing actions. Downstream from several project sites it becomes increasingly difficult to attribute an in-channel habitat condition trend to an individual action but this trend does reflect effectiveness of the entire management program. It may be feasible to find relatively pristine small headwater basins to act as a controls for other small, managed basins, but comparisons of habitat conditions must follow a well-conceived classification system.

Level 1 monitoring is focused on the core habitat conditions shaping salmon survival that are identified in the CSP (Rhodes et al. 1994). This monitoring level provides the data upon which important management decisions are made when it has been decided that the cumulative effects of land management must be managed to provide assurance of improvement in in-channel habitat conditions upon which listed fish species depend. Part of this monitoring effort must also be directed to **implementation** monitoring. This is basically an accounting activity to ensure that an approved management program is carried out as planned, to keep track of how much of what kinds of activities were conducted, and to map out where they occurred.

The Clearwater drainage is a large watershed within the Snake River subbasin (Fig. 1) comprised of three major tributaries, the North, Middle, and South Forks. This drainage provided several case studies of federal watershed management and its long-term effects on salmon habitat (Rhodes et al. 1984). This river system also provides a convenient means to illustrate the terminology pertinent to watershed hierarchy used in this document. The Lochsa and Selway Rivers are primary tributaries that join to form the Middle Fork Clearwater. Each is approximately 6th order. The Lochsa River constitutes a significant salmon production system (Fig. 2). Its secondary tributaries such as Fish, Hungry, Pete King, Squaw, Weir, Post Office, Papoose, Crooked Fork, Brushy Fork, White Sand, and others are 4th-5th order "secondary tributaries" (see Table 1) that each contribute salmon spawning and rearing area. The mainstem Lochsa River between Post Office Creek and the mouth of White Sand Creek also provides significant spawning area. The upper Lochsa River provides a major portion of the total salmon production of the entire system. It has three major salmon-bearing tributaries, Brushy Fork, Crooked Fork, and White Sand Creek (Fig. 3). Brushy Fork and Crooked Fork have suffered major management impacts over past decades that have seriously damaged the current productive capacity. White Sand Creek has a large portion of its watershed in a wilderness area, which gives its habitat system high quality and stability. However, the Clearwater National Forest is planning to harvest timber from much of the watershed under the guise of ecosystem management. Should this happen, this monitoring framework would provide guidance for assessing the extent of damage to ensue. For example, a contrast of stream reaches in wilderness versus the managed watershed, as well as trends at the managed site, may demonstrate the negative impacts of management.

As a brief sketch of the monitoring framework, Level 1 analysis will provide a status report on the entire Lochsa salmon producing system, as well as that of each of its individual salmon-bearing tributaries. Data must be available on the current spatial distribution of salmon production, the overall condition of each stream system, and the condition of the riparian system for each salmon-bearing drainage. Monitoring will emphasize the variables for which standards were recommended in Rhodes et al. (1994). Data must be available at this coarse level resolution for the entire drainage so that management decisions can be made effectively when they have the potential to adversely affect the productive capacity of the entire system. For example, to take actions that are likely to increase fine sediment levels in White Sand Creek, a major remaining salmon production watershed, when the majority of the rest of the Lochsa salmon habitat has been damaged and the dependent salmon populations are on continuing downward trends, would be biologically unsound.

Level 2 analysis will serve as the "early warning" device for detecting threats to downstream salmon habitat. It requires monitoring of 0-3rd order tributaries within or adjacent to the project as well as the site of management actions. Fixed stream reaches monitored over time establish trends relative to management history. Effects far downstream in the important spawning reaches of the mainstem Lochsa River (Fig. 3) are produced from cumulative management actions in all upstream, managed non-salmon-bearing tributaries. These reaches are slow to respond to individual upstream actions, but are also resistant to improvement unless coordinated, geographically widespread improvements in management (e.g., application of the CSP management guidelines) are applied consistently.

Figure 1. The Clearwater drainage as a large watershed within the Snake River subbasin.

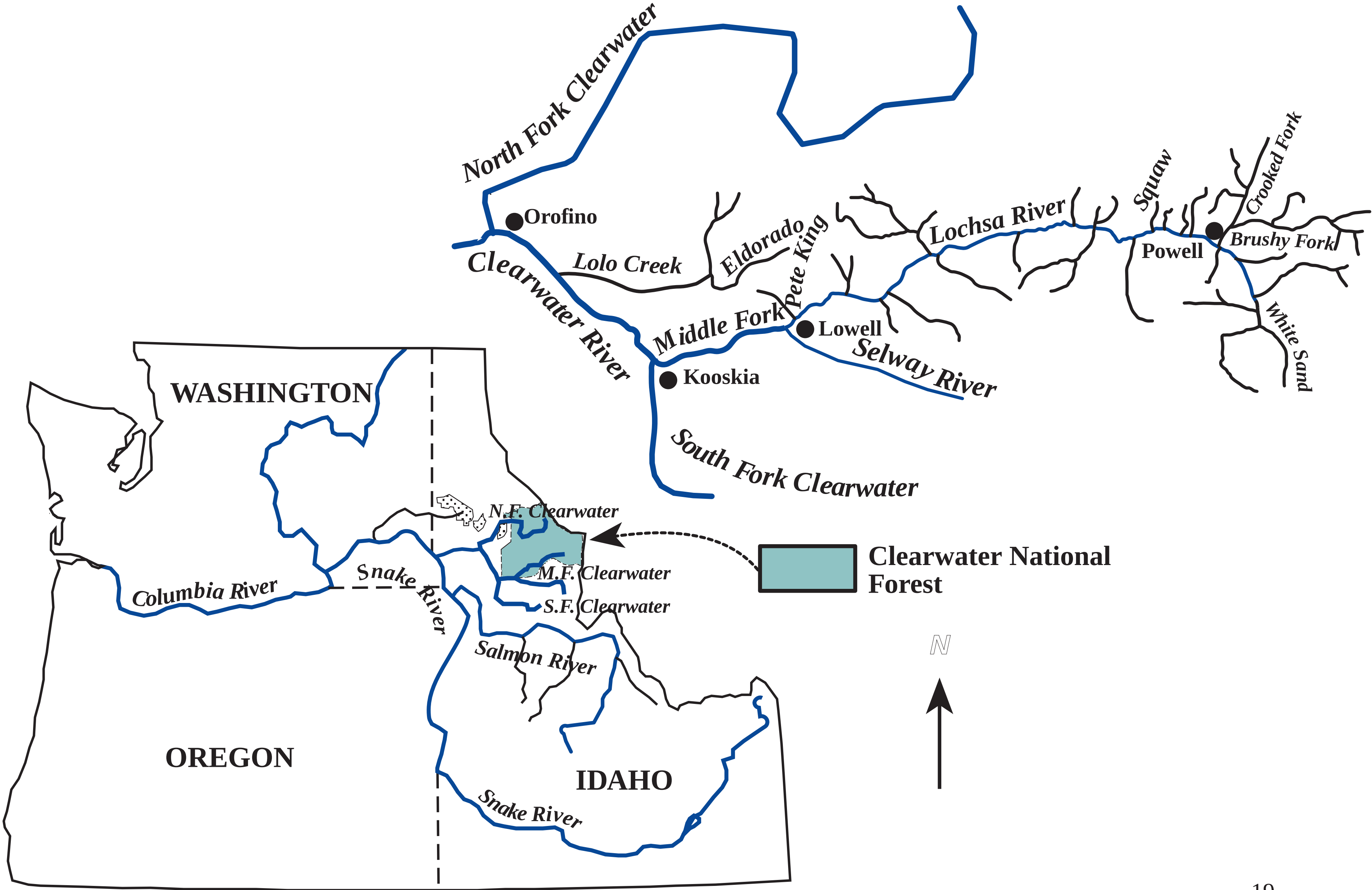
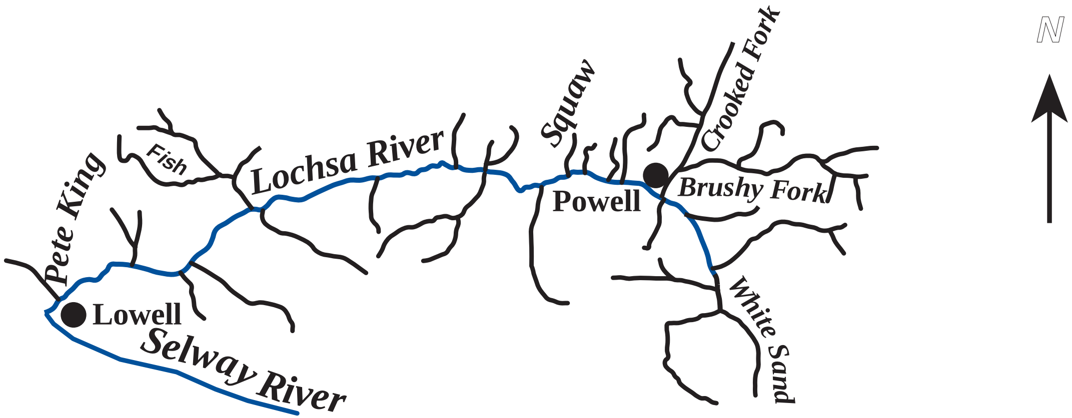
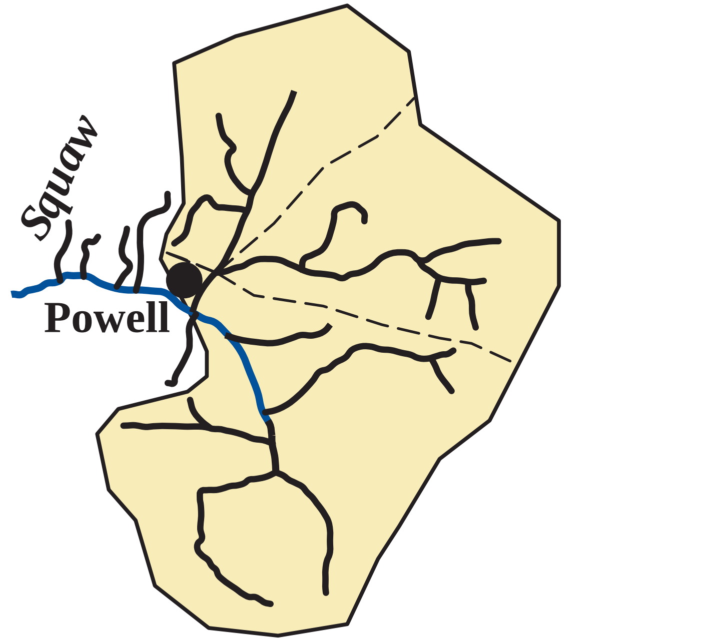


Figure 2. The Lochsa and Selway drainages as primary tributaries to the Clearwater River.

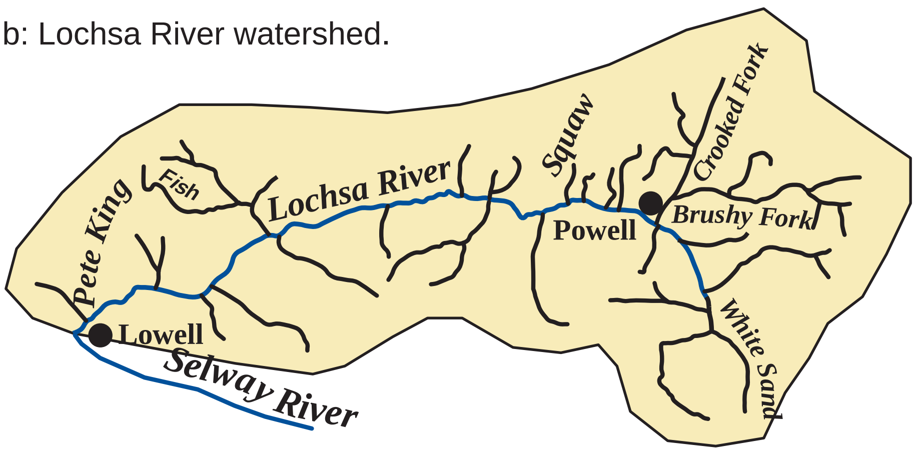
a: Lochsa River with some of its salmon- and non-salmon bearing tributaries.



c: upper Lochsa River and three major salmon-bearing watersheds.



b: Lochsa River watershed.



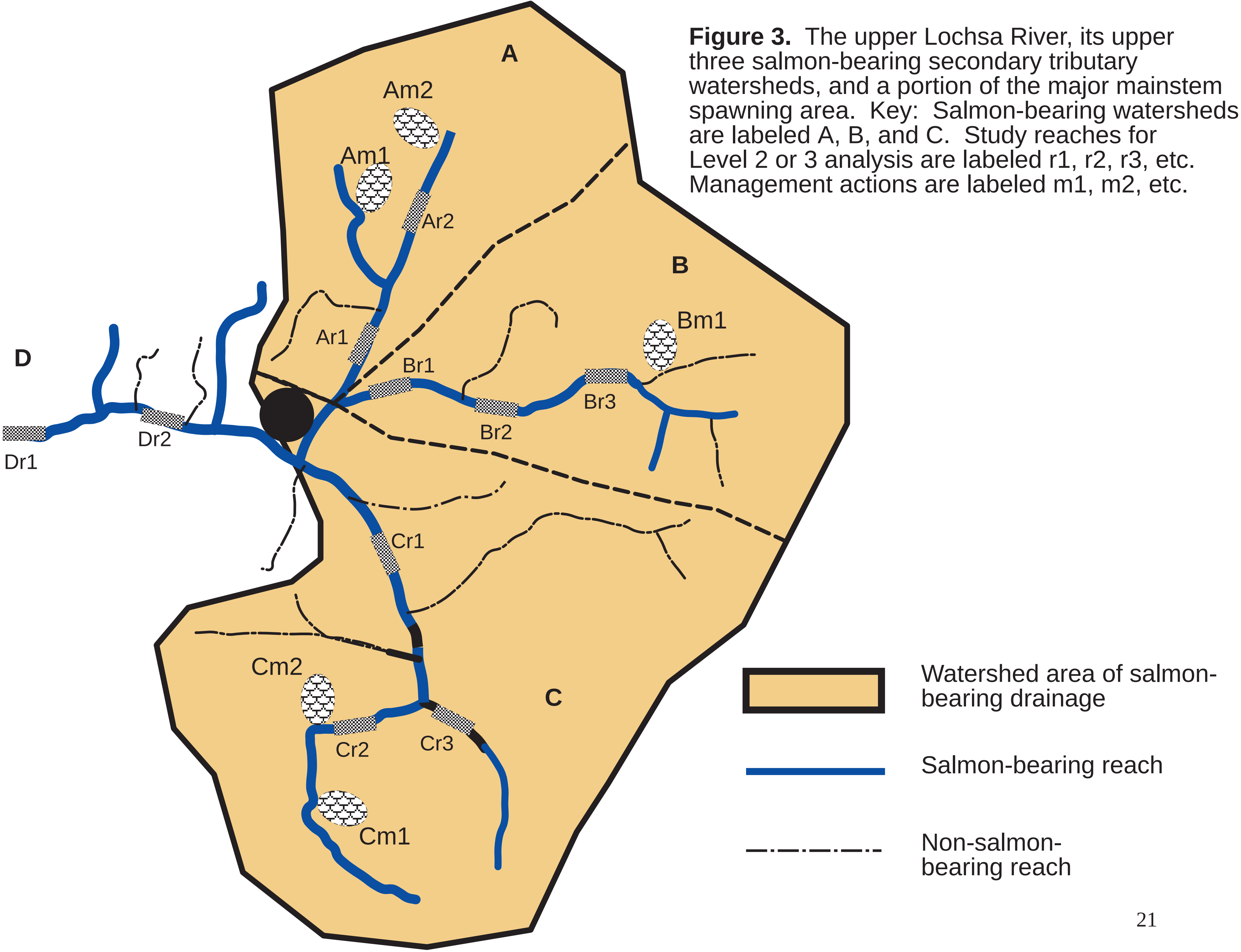


Figure 3. The upper Lochsa River, its upper three salmon-bearing secondary tributary watersheds, and a portion of the major mainstem spawning area. Key: Salmon-bearing watersheds are labeled A, B, and C. Study reaches for Level 2 or 3 analysis are labeled r1, r2, r3, etc. Management actions are labeled m1, m2, etc.

Part III: Objectives of Monitoring at Each Level and the Analyses to be Performed

Level 1. Extensive Environmental Assessment: Salmon

Geographic Unit for Analysis

Salmon-bearing watersheds from the scale of secondary or primary tributary to sub-basin level. Example progression of watershed scales: White Sand Creek→ Lochsa River→ Clearwater River→ Snake River.

General description

Monitoring consists of evaluation of watershed and salmon habitat condition using CSP variables (see Appendices B, C, and D) as a minimum set for a watershed of a secondary tributary (4th to 5th order), all watersheds within the sub-basin (a geographical array), or the sub-basin as a whole (by aggregation of individual watershed analyses).

It will emphasize:

A synoptic view of current condition and trends of key in-channel habitat and watershed parameters and attainment of standards expressed in the CSP. Data collected using this coarse level monitoring protocol provide the essential data that allow management decisions to be made in watersheds supporting ESA-listed fish stocks.

Screening the implementation of CSP land use standards.

Geographic aggregation to assess geographical robustness of recovery as measured by habitat and population condition and trends in relation to trends in condition of the watershed. Habitat trends along a river continuum are evaluated using multiple variables as a response to cumulative effects to land management actions and spatial distribution of current condition by site.

Display and evaluation of cumulative effects down a river continuum.

Population trends in fish species of special concern and trends in the condition of their entire habitat system (spatially organized habitat units) and all spawning and rearing habitat.

Objectives

Level 1 monitoring for salmon-bearing watersheds is geared to provide **synoptic analysis** (or **watershed reconnaissance**) of total system condition. It is designed to assess the total quantity and quality of salmon spawning and rearing habitat; the condition of the entire riparian system; and watershed condition at the watershed scale and trends over time. The monitoring data provided are essential to the Coarse Screening Process. However, this information is also vital to adaptive management and monitoring for ESA goals and legal requirements (Henjum et al. 1994, Rhodes et al. 1994). The objective of Level 1 monitoring is to provide evidence regarding the **baseline condition and trend** in key habitat parameters on a system-wide basis. As a screening tool, this survey summary is needed to evaluate whether restoration is needed, low risk land management activities should proceed, and efforts to improve and protect habitat are successful over time. Following the flow chart of the CSP monitoring protocol (Rhodes et al. 1994, as amended by R. Beaty; see Appendix D), one would conclude that if data are not available concerning the key habitat parameters or if these data show that standards are not being met on the drainage in question, then activities that could potentially forestall recovery (see Table 2) should not be continued or implemented until there has been a 5-year improving trend, standards are met, or >90% of managed watersheds improve.

The Level 1 (coarse scale) view of the entire system is achieved logistically by statistical sampling procedures [e.g., Hankin and Reeves (1988) or the Dolloff et al. (1993) Basinwide Visual Estimation Technique] and by total inventories by use of remote sensing (aerial photo, satellite imagery analysis, etc.) to gain synoptic overviews of system health. Coarse scale mapping of habitat quality (using the screening parameters as a base data set) is best summarized by loading on a GIS. Data from statistical sampling can by used to derive area-weighted means for various channel types comprising the stream system. This overview then allows representative portions of the system (watershed, riparian, channel types, channel units) or strategic points to be selected so that (1) more precise estimates of condition and trend can be achieved, (2) these estimates can be contrasted with the interpretation made at the coarse scale and used to calibrate coarse estimates by channel type, (3) and the utility of coarse screen parameters can be assessed by cross referencing them to estimates made using more precise techniques or techniques designed to measure other related factors.

Monitoring at the watershed scale to assess conditions in logical salmon production units involves evaluation of 4th-5th order watersheds, at minimum. The analytical unit for purposes of CSP monitoring is either the entire stream network or total salmon spawning or rearing habitat for in-channel parameters and is the entire watershed, riparian, or floodplain system relative to land use parameters of the CSP. When considering CSP in-channel parameters for substrate condition, one would evaluate percentage surface fine sediment in spawning areas and cobble embeddedness in rearing areas. It can be evaluated using either total sampling (e.g., visual evaluation of the entire habitat class, such as the total current or historic spawning habitat; counting all LWD in the habitat class or salmon-bearing reaches), by sub-sampling the total salmon-bearing stream reaches or separate channel types or habitat classes (e.g., using Hankin and Reeves, transect sampling). Evaluation of water temperature should be focused in current or historic salmon-bearing reaches. Violations of the temperature standard detected at any reach should prompt evaluation of the longitudinal temperature profile, riparian cover, and channel width within a zone of influence (e.g., 25 miles upstream from the violation) (Rhodes et al. 1994). Application of a reach temperature model using riparian condition data, aerial photo analysis of riparian condition trends, and records of historic conditions may indicate the magnitude and rate of potential improvement in temperature conditions that could be achieved by restoration. Recovery of the temperature profile for the entire reach should be interpreted with respect to the recovery of shade and channel structure.

Bank stability is a CSP in-channel habitat parameter that should be evaluated for the entire stream network of the salmon bearing watershed. It should be evaluated by total sampling making visual estimates of a series of 20-m reaches. Total sampling should be feasible for all 3rd-5th order reaches; however, for 1st-2nd order reaches it may be more reasonable to create a subsampling process. This should preferably be based upon stratification of these small watersheds by landtype, land use, or classes of riparian impact visually determined from aerial photos.

Other in-channel habitat parameters, suggested in the CSP as auxiliary parameters, should be monitored in salmon habitat. Provided that land use standards suggested in the CSP are in place, monitoring of these auxiliary parameters may not be necessary for screening purposes. However, they serve other needs. They provide corroborative information regarding key screening parameters. For example, the assumption that monitoring of surface fine sediment percentage reflects fines by depth must be validated. In addition, monitoring variables such as LWD, pool frequency and volume, residual pool volume and stream shading allow improving or declining trends to be detected and factored into management. Declining trends in these important parameters should be considered in conjunction with current state of CSP screening parameters in recommending increased levels of protection and restoration (e.g., road obliteration to reduce sediment delivery) or necessary modification in land use practices.

Level 1 monitoring of watershed and in-channel habitat condition should include GIS mapping of all streams within the watershed, subwatershed boundaries, road system, wetlands, riparian reserves, roadless reserves, mining sites, allotments, areas of mass failure potential and occurrence, etc. Aerial and ground photos constitute a major source of data for historical documentation of channel and riparian condition and change. These data on stream system change are best interpreted against historical discharge records, including base and peak flows This map base will provide the analytical tool for calculating or identifying existence of roadless areas, riparian reserves, the amount of road mileage, the extent and location of harvest units, the extent and location of chronic sediment sources, the location and size of wetlands and water withdrawals, the location and magnitude of point sources of pollution (mining projects), the location, amount, and intensity of riparian harvesting and grazing, and stratification of the land base into land use allocations or capability units according to an existing Forest or Land Use Plan. This information will establish the general condition of the watershed plus specifically identify problem areas and watershed sites requiring restoration. A basin-wide survey of sediment sources is a necessary precursor to improving in-channel habitat conditions in sediment-affected channels (Kondolf and Larson 1995).

Total spawning and rearing area and areas with critical spawning or rearing habitats should be identified and mapped. This reconnaissance information is important for establishing representative monitoring sites for Level 2 monitoring as well as for rapidly identifying threats to unique components of habitat.

Upon the hierarchical framework of watersheds presented in Table 1, GIS mapping of distribution and condition of ecosystem types (wetland, riparian), management units (landtype, roadless reserve), potential vegetation type, soil type, geology, Rosgen (1994) channel type, valley type, and climate (precipitation, snowpack, temperature) should be overlaid to facilitate analysis of in-channel habitat condition relative to the contributing watershed or estimates of watershed sediment delivery. Baseline and trend data on current condition of these ecosystem or management units must be developed in Level 1 monitoring. This should include monitoring of ecological and stand condition of vegetation in the allotments (upland and riparian areas), wetlands, riparian areas, landtypes (considering harvest units of varying ages and unharvested stands). In addition, information on streamflow statistics and channel cross-sections at each monitoring site is an essential aspect of assessing hydrological character of the sites. Hydrological characteristics can be effectively used to explain biotic distribution in streams. High or low flow levels are frequently associated with salmon escapement, survival, spawning area available, etc. Streamflow measurements are useful in interpreting data related to flow (substrate condition, temperature) and are essential in calculations of total transport (e.g., sediment load, nutrient output). It is important to provide long-term streamflow gages to augment USGS records and so that measured flows at monitoring sites can be correlated with nearby permanent gages so that flows can be extrapolated from gaged to ungaged sites.

In addition to the synoptic analysis provided for purposes of the CSP, Level 1 monitoring should provide **implementation monitoring**. Implementation monitoring has as its objective determining whether the land management actions planned (i.e., BMPs, contract provisions, restoration actions) are actually carried out. This frequently identifies little more than whether the estimated number of board feet are removed, the planned livestock AUMs correspond to actual AUMs, miles of road to be constructed/reconstructed were accomplished, or the prescribed number of acres to be treated (burned, planted, etc.) occurred. Past experience has also demonstrated that contract provisions are frequently violated. For example, riparian buffers often are reduced in width during logging despite clear directions to leave wider buffers. Livestock are frequently allowed to graze allotments far past the prescribed grazing season. Implementation monitoring requires careful documentation of the degree of compliance of contractors and permittees.

Federal land managers also need to assess the degree of implementation of program actions their staff take. For example, if all culverts are to be inspected on an annual basis for proper operation but they are inspected every three years because the road system has expanded beyond the ability of the district to provide inspection, the threat to the resource from limited implementation monitoring must be documented. A consequence of low frequency culvert inspection is an increased incidence of tributary diversion across road fills and increased magnitude of erosion (see Hagans and Weaver 1987).

It cannot be assumed that BMPs, applied to individual projects, are effective or sufficient, especially when several BMPs are employed simultaneously or in sequence or when the total level of activity in a watershed (i.e., number of individual projects) is not regulated (see Rice 1992). Therefore, it is critical that BMPs (whether they be CSP land use standards or other land management standards) are implemented as planned prior to determining their effectiveness in maintenance or improvement of stream conditions. Even with adoption of CSP land use standards, monitoring of their implementation and effectiveness is required. Also implementation monitoring of activities passing the screening process and recommended restoration actions is needed.

It is recommended that a full accounting of all implementation monitoring be provided. This should indicate the intensity and spatial distribution of sampling (representativeness of the action monitored) and the area monitored in relation to the area affected by the entire resource program. That is, it is important to know whether all implementation monitoring is concentrated in one part of the forest or subbasin, whether only very spatially insignificant actions are monitored, and whether all aspects of contract compliance are monitored. Disposition of cases of non-compliance should be reported. That is, if logging companies or livestock permittees violate contracts, the actions taken must be reported. Plans for follow-up monitoring of problem sites and compensatory modifications to the program should be identified. The current condition of habitat should be available as a GIS database for entire sub-basins. Similar databases linked to GIS should be available to provide an overview of implementation and effectiveness monitoring.

The purpose of implementation monitoring is to act as a first line in the defense of habitat integrity. If management actions are seldom carried out as planned and are not well documented, it becomes difficult to link land management actions with stream channel condition. It also becomes difficult to determine the effectiveness of land use prescriptions unless it can be documented that existing prescriptions were actually carried out.

Effectiveness of the total land management program (e.g., achieved via implementation of recommendations in Rhodes et al. 1994) in protecting and restoring salmon habitat can be assessed at coarse resolution from general trends in CSP screening parameters and other recommended parameters for habitat condition. Such a monitoring program makes use of long-term trend data from the synoptic surveys.

Approach

a. Essential elements

- Gather/collect available data on land use, landscape characteristics, watershed condition, redd counts, fish escapement (reports, data, maps).
- Use aerial or satellite photo analysis.
- Gather/conduct fish habitat surveys on all tributary watersheds.
- Enter data into GIS so that overviews of watershed condition can be produced at scales of secondary tributaries to the subbasin.
- Synoptic view of current condition for the primary tributary can be composed by compilation of all analyses done on headwater and secondary tributaries plus the continuum of mainstem habitat and associated riparian reserve condition. Absent the intensive data by headwater and secondary tributary, perform analysis at a coarser scale of resolution.
- Identify and map riparian community types.

b. Optional elements.

Useful in order to extrapolate results to other similar ecological situations; to derive replicates having similar capabilities.

- Identify and map ecoregions (Bailey 1976, 1980; Omernik and Gallant 1986)
- Identify and map landtypes (Wertz and Arnold 1972)

- Identify and map channel types (Rosgen 1985, 1994)
- Identify and map channel unit types (Bisson et al. 1982; Hawkins et al. 1993)

Key issues to be addressed

a. In-channel conditions. Are in-channel habitat standards and desired trends in habitat condition met in watersheds within the sub-basin?

Habitat conditions involved in monitoring:

> *1. Variables for which numeric standards are set:* monitored for current condition to determine compliance; monitored to establish trends to determine effectiveness of land management in protecting and restoring salmon habitat.
>
> - surface fine sediment
> - cobble embeddedness
> - bank stability
> - water temperature
> - water quality
>
> *Are state and federal water quality laws being enforced?*
>
> > Review state water quality reports submitted to EPA or the Corps of Engineers. Compile existing data from state agencies (Department of Fisheries, DEQ, DNR, Dept. of Water Resources, Dept. of Forestry) and federal agencies (USFS, BLM, USGS, NRCS, BR) on stream water temperatures. Compare with state and agency standards. Map via GIS the stream zones exhibiting water quality standards violations (temperature, dissolved oxygen, pH, bacteria, turbidity, nitrate, etc.).
>
> *2. Variables for which no numeric standards are set:* monitored for current condition to assess baseline; monitored as trend from current baseline to detect trends and determine effectiveness of land management in protecting/restoring habitat.
>
> - large woody debris frequency and volume
> - pool frequency and volume
> - residual pool volume
> - stream shading
> - water quantity and timing

b. Watershed. Are land use standards being implemented? If they are implemented in management plans, what land allocations are created? What are the current activities by land unit; what is the current condition and trend for these land units?

> *1. Has the riparian reserve system been established?*
>
> The purpose of this reserve is to let natural restorative processes operate without perturbation

from anthropogenic sources. These ecological processes should aid in re-emergence of historic, high quality habitat conditions and avoid incremental or on-going habitat damage. Monitoring must first assess whether riparian systems have been allocated to management as a reserve (i.e., whether this provision of the CSP was implemented).

Determine by implementation monitoring.
Procedure for synoptic evaluation: review all implementation reports; summarize reports. Determine the extent of the riparian reserve system; map it using GIS.

What is the current state and trend in riparian condition? Has the integrity of the riparian reserve system been violated?

Conditions involved in monitoring: review of management plans; status and trend reports from aerial or satellite imagery on condition and extent of the riparian system and road density; impact of mining in the riparian reserve; amount grazed by livestock and ecological status; aggregation of field mapping of riparian stands.

2. Has the roadless reserve system been established?

The purpose of the reserve system is to act as a high quality refuge for native species, including the listed species. It also serves as a naturally functioning ecosystem where processes contribute to habitat maintenance and as the anchor point for habitat restoration efforts. Such systems are also critical as reference points for monitoring ecological communities and processes.

Determine by implementation monitoring.
Procedure for synoptic evaluation: review all implementation reports; summarize reports. Determine the extent of the roadless reserve system; map it using GIS.

What is the current state and trend in roadless reserve condition? Has the integrity of the roadless reserve system been violated?

Conditions involved in monitoring: status and trend reports from aerial or satellite imagery on condition and extent of the roadless reserve system; aggregation of field mapping of roadless tracts; evidence of road extension; evidence of threat to the roadless reserve from management practices in adjoining lands, such as disease spread, fire suppression or spread from slash burning, blowdown threat to the reserve perimeter by excessive logging up to the reserve; is water being withdrawn for irrigation.
Are land use standards being applied to activities in the riparian zone and floodplain?
If the bank condition for any reach of at least 100 m is <90% stable, are ground-disturbing activities being allowed in the riparian zone and within ½ site potential tree height of the floodplain?
If the water temperature standard is not met, are activities (shade-removing or ground-disturbing) occurring in the riparian zone?

3. Do sediment delivery estimates for all combined sources indicate that sediment delivery in salmon-bearing watersheds is ≤20% over natural background?

Apply the appropriate variant of the R1-R4 sediment model. The model should be one that is updated to include grazing, all existing roads, and mining.

The purpose of estimating total sediment delivery from watershed activities is to monitor trends in levels of combined sediment producing activities that contribute to habitat degradation. Rhodes et al. (1994) (see their Table 2) specify that if sediment delivery is >20% over natural, no land disturbing activities should occur until sediment delivery is reduced to <20% and in-channel substrate conditions meet standards. Monitor to determine what kinds of land disturbing activities are taking place. A structured system for estimating these cumulative impacts is important so that rational decisions can be made regarding actions for improving salmon habitat substrate conditions.

Environmental conditions involved in monitoring: current road density; miles of road obliterated or relocated on the watershed and in the riparian reserve; livestock density and grazing systems employed by land type; acres of historic timber harvest; proximity of agricultural land, acreage, and type of agriculture; distribution of sensitive hillslopes; extent and type of mining activities.

4. What is the current distribution and extent of wetlands? Are wetlands being lost? Is wetland area within the sub-basin or watershed less than occurred historically?

The preservation and restoration of wetlands, wet meadows, and floodplains is essential in maintenance and restoration of watershed function. It provides cold, stable water sources, significant fish rearing areas, and sediment storage sites.

Environmental conditions involved in monitoring: historic and current wetland area and distribution. Identify vegetative species for wetland communities by land type.

Identify opportunities to restore wetlands, wet meadows, and floodplains.

5. Is surface water or groundwater being withdrawn to the detriment of fish production? Are current holders of water rights using more than allowed amounts of water?

Drought during the summer period is a periodic problem whose effect on magnitude and frequency of streamflow is exacerbated by additional water removal or shifts in timing of flows. Lower flows can cause passage blockages, aggravate water temperature problems (increase maximum daily temperature or cause seasonal delays in temperature peaks in mainstem reservoirs), and can reduce effective rearing or spawning area.

Environmental conditions involved in monitoring: water flow in irrigation systems or volumes pumped; hydrologic connection between surface flow and groundwater.

Are new water rights being issued?

Environmental conditions involved in monitoring: reports from water resources departments.

Confirm that all additional withdrawals are denied until it can be ensured that spawning and rearing flows are adequate on a subbasin scale.

D*oes the existing water flow and timing fully meet the biological needs of salmon?*

Study instream flow needs of salmon in an effort to support acquisition of sufficient flows.

Identify opportunities to acquire rights to water for maintenance of instream flow needs of salmon.

6. Are fish being lost in the irrigation systems?

Environmental conditions involved in monitoring: incidence of lack or malfunction of diversion screens; counts of fish trapped in irrigation system.

7. Does handling, transport, and storage of toxic material conform to recommended guidelines?

Monitor the uses of toxic materials (e.g., cyanide, diesel fuel, acids or bases, pesticides/herbicides) in watersheds supporting spawning and rearing.

8. What are the opportunities to undertake active restoration action?

Environmental conditions involved in monitoring: roaded area, location, quality, and geologic setting; occurrence of underfit or impassable culverts; unstable hillslopes; soil and vegetation disturbance leading to erosion and poor ecological condition; opportunities to acquire water rights or property to add to a roadless or riparian reserve system or to restore a wetland.

c. Biotic indices

1. Population trends in salmon. What do the salmon redd count trends for all watersheds within the sub-basin indicate about the species/stock status? What are the trends in abundance and composition of the anadromous fish community; the listed species?

Habitat condition: air temperature regime as it affects snowmelt rate and water temperature; water discharge regime, water depth, turbidity, presence of intermittent passage barriers downstream.

Land use condition: factors affecting water quantity and timing such as extent of clearcuts; irrigation diversion,
Biotic indices: redd counts/mile for all anadromous fish. Contrast trends for salmon with steelhead.

Level 1. Extensive Environmental Assessment: Non-Salmon

Geographic Unit for Analysis

Non-salmon-bearing headwater tributary to secondary tributary level. example scale: Mex Creek to White Sand Creek

General description

The extensive environmental assessment and inventory involves evaluation of watershed and headwater tributary channel condition using selected CSP variables as a minimum set. Screening is not conducted at this scale but is done at the level of the 4th to 5th order salmon-bearing watershed, a level that incorporates its non-salmon-bearing watersheds.

Focus on early detection of watershed and stream channel problems arising from forest management as an early-warning to cumulative downstream effects.

Determine whether channel conditions indicate adverse levels of watershed perturbation.

Feedback from headwater tributary monitoring is essential for averting effects in salmon habitat and for identifying restoration opportunities. It also increases the ability to make interpretations about causative mechanisms responsible for cumulative downstream effects.

Biotic monitoring provides an information base on status and trends in resident fish, macroinvertebrates, and riparian vegetation species and communities. Biotic indices for species and communities associated with channel types or channel units reflect long-term trends in numerous watershed or habitat condition parameters.

Objectives

Level 1 monitoring for non-salmon-bearing watersheds is directed at assessing the conditions and trends of headwater tributary watersheds and their major components (e.g., riparian, floodplain, stream channel). Monitoring of in-channel and watershed condition and trend in the small non-salmon-bearing watersheds provides data that can be aggregated to the scale of the salmon-bearing watershed for implementation of the CSP. The watershed-wide monitoring of these low order watersheds is intended to both provide a portion of the necessary data for aggregation and analysis at the 4th to 5th order (salmon-bearing watershed) level and to serve as an "early warning device" at the site of many forest-based activities to allow rapid feedback and prevention of downstream degradation.

Monitoring involves a combination of assessing implementation of planned actions (including CSP land management standards), reconnaissance inventory of condition of the watershed, riparian zone, and in-channel habitat conditions, including indications of sediment and temperature problems. Just as for monitoring at the 4th-5th order watershed scale, if recommended CSP land use standards are in place, monitoring of in-channel habitat parameters such as LWD will not be essential as a means to provide feedback for impacts of riparian management. In this case, establishment of riparian reserves and implementation of riparian revegetation actions will lead to eventual expression of late-successional LWD loading volumes. However, monitoring of LWD, pool quality and frequency, and other parameters recommended as indicators of condition and trend in the CSP (Rhodes et al. 1994) provides information on habitat recovery success with implementation of CSP land use standards.

Approach

- Gather available data on land use, landscape characteristics, watershed condition.
- Use aerial or satellite photo analysis plus ground-truthing or extensive ground-based surveys of the watershed or stream to map problem management situations.
- Trace sources of apparent degradation in salmon-bearing streams to headwater tributaries if necessary.
- Identify and map current riparian community types. Based on landforms, elevations, soils, valley type, map inferred late successional riparian community types.

- Identify and map ecoregions (Bailey 1976, 1980; Omernik and Gallant 1986)
- Identify and map landtypes (Wertz and Arnold 1972)
- Identify and map channel types (Rosgen 1985, 1994)
- Identify and map channel unit types (Bisson et al. 1982; Hawkins et al. 1993)

Key issues to be addressed

a. In-channel. Are in-channel habitat standards being met?

Habitat conditions involved in monitoring:

Measure these variables according to longitudinal distribution (i.e., along the river continuum) of their values, and with respect to channel type and channel unit. For example, the downstream temperature profile, accompanied by a map of groundwater and tributary water entry points (volume and temperature of water additions), provides a broad view to the state of the riparian system and alerts one to problem locations where abrupt deviations occur or greater than average rates of stream warming occur.

1. Variables used as numeric standards; monitored for current condition to determine compliance with biologically-based standards; monitored as trends to determine rate of restoration; effectiveness of land use prescriptions as a whole in a watershed; effectiveness of single activities carried out locally.

- surface fine sediment
- cobble embeddedness
- bank stability
- water temperature
- water quality, including turbidity and suspended sediment

2. Variables for which no numeric standards are set; monitored for current condition to determine baseline; monitored as trend from current baseline to detect restoration or worsening condition.

- large woody debris
- pool frequency and volume
- residual pool volume
- stream shading
- water quantity and timing

3. Variables for which no numeric standards are set but which can provide extra information in headwater tributary extensive assessment; monitored as trend from current baseline to detect restoration or worsening condition.

- sediment storage, measured as wedges upstream of LWD; or bar or floodplain deposits measured by ground-based, aerial mapping, or extensive transects.

b. Watershed. Are land use standards being implemented?

Apply the same procedures as for the salmon-bearing tributaries (see p. 27)

c. Biotic indices. What do biotic indices reveal about the status and trends in habitat quality or progress in restoring the watershed?

1. What is the status and trend in abundance and composition of the resident fish community?

Biotic and environmental conditions involved in monitoring:

- species composition.
- distribution of species with respect to channel type, channel unit type, position in river continuum.
- age composition of key species.
- presence of exotic species.
- presence/absence and/or numerical abundance of key species by age class.

Classification: capacity—channel types, channel unit types, and position within river continuum (incorporating elevation), landtypes; performance—current water temperature regime, riparian composition and cover, substrate composition, bank stability, pool quality and frequency.

2. Do anadromous fish use streams or stream reaches previously considered to be resident fish only or non-fish-bearing? Does the use of these streams or reaches by anadromous fish change over time with restoration?

Tributaries are often misclassified according to fish use. This leads to application of inappropriate state forestry standards. State standards are typically unprotective because they subdivide an integrated salmon production system into fish and non-fish-bearing tributaries and apply different management to them as if they were independent. At a minimum, salmon-bearing tributaries should be classified correctly. In addition, water temperature, pool, substrate, and bank restoration allows a greater extent of each tributary to serve as habitat.

Biotic and environmental conditions involved in monitoring:

- species composition
- distribution of species with respect to channel type (incorporating channel gradient and W/D), channel unit type, position in river continuum
- age composition of key species
- presence of exotic species
- presence/absence and/or numerical abundance of species by age class

Classification: capacity—channel types, channel unit types, and position within river continuum (incorporating elevation), landtypes; performance—current water temperature regime, riparian composition and cover, substrate composition, bank stability, pool quality and frequency.

3. What is the status and trend in abundance and composition of the macroinvertebrate community?

Although it could be considered sufficient to monitor a few variables such as percentage fine sediment or water temperature as indicators of ecological health and endpoints in stream recovery, biotic monitoring, including macroinvertebrate monitoring, provides important confirmation of the adequacy of whatever variables are measured. Because the measured and numerous other unmeasured variables all interact and can produce synergistic or cumulative effects that cannot be fully predicted, it is important to conduct biotic monitoring, especially of species that are subjected for long periods to these multiple effects. Macroinvertebrates are the food base upon which salmon and resident fish depend. Experience gained in monitoring freshwater biotic distribution over broad geographic scales and stream orders for each region will become increasingly useful in establishing biotic reference standards and future potential indicators of broad geographic recovery or collapse.

Biotic and environmental conditions involved in monitoring:

- species composition of the invertebrate community; relative density and biomass of taxa using kick sampling or cobble surface sampling
- distribution and relative abundance of species with respect to channel type, channel unit type, position in river continuum
- assess biotic status for specific channel types, channel unit types, and position within river continuum employing a rapid bioassessment protocol (e.g., Fore et al. 1995)
- relative abundance of sediment or temperature sensitive species

Classification: by capacity—channel types, channel unit types, position within river continuum (incorporating elevation), and landtypes; by performance—current water temperature regime, riparian composition and cover, substrate composition, bank stability, pool quality and frequency.
Assess density and biomass according to functional feeding groups; family and order taxonomic units; tolerance of temperature, fine sediment, and flood disturbance.

4. What is the status and trend in abundance and composition of the riparian community.

Broad geographic scale distribution of riparian vegetation (current, historic, potential natural vegetation) should be mapped to establish zonation of riparian communities. This may eventually serve as a guide to most appropriate species to reestablish in riparian reserves when active restoration is needed. It will also record progress in ecological restoration of riparian zones that can serve as models for other similar sites. It could provide a broad restoration strategy to be applied extensively in a watershed. This might include development of general expectations of riparian cover type and density by altitude.

- dominant species by cover class (tree, shrub, groundcover)
- percent canopy cover
- height class
- density of dominant tree species

Classification: channel types, channel unit types, and position within river continuum (elevation), landform, soil type.

Level 2. Intensive Trend Evaluation: Salmon

Geographic Unit for Analysis

Salmon-bearing watersheds from the scale of secondary or primary tributary to sub-basin level. Example progression of streams: Fish Creek or White Sand Creek, Lochsa River, Clearwater River.

General description

Intensive trend monitoring focusing on seasonal abundance and species composition of key salmonids and habitat quality in critical spawning and rearing sites, using the same CSP variables as in Level 1 plus additional variables as required.

General determination of effectiveness of land use practices in improving adjacent or downstream habitat conditions as expressed in trends in selected salmon habitat types and locations.

Detailed mapping of physical structure and condition of in-channel habitat and associated riparian, floodplain, and sideslope area.

Objectives

Level 2 monitoring for salmon-bearing watersheds is intended to provide intensive documentation of habitat condition and trend and salmon populations in representative portions of salmon-bearing streams. Many of the same kinds of habitat parameters that are of importance in Level 1 monitoring are also emphasized in this more intensive monitoring level. In-channel sample sites monitored intensively must be selected strategically to reflect overall trends in watershed, riparian system, or local riparian or sideslope condition. It is virtually impossible to establish experimental sites with perfect controls for comparison in the field. However, use of physical classification procedures allows sites of similar type to be stratified so that in-channel conditions can be rationally explained in terms of their immediate or extended (i.e., total watershed) environment. A coarse longitudinal stratification (e.g., Hankin and Reeves 1988) can be used to identify upper, middle, and lower reaches of the test and control streams (i.e., streams within watersheds of a given class that range in habitat condition due to level of management impact). Otherwise, the stream continuum can be segmented using other geomorphically-based stratification procedures. One useful classification of this type is Rosgen's (1985, 1994) channel type classification. Representative (especially, dominant types) and sensitive channel types that are relatively homogenous within these reaches should be selected for monitoring.

Level 2 analysis of salmon-bearing streams using CSP parameters will be feasible using Level 1 data provided that Level 1 data are coded by all relevant component hierarchical levels (e.g., channel type, channel unit, etc.) for the specific reaches rather than aggregated at the watershed level. These data would also need to be obtained at the same level of precision as appropriate for analysis at Level 2. If Level 1 data are of coarse resolution but extensive, they will serve the function of establishing a general monitoring framework or representative sites for Level 2. Level 2 analysis should include a complete mapping of the channel at the selected monitoring sites and the distribution of channel units within them. This will provide information on length, width, and area of each channel unit. For each channel unit critical features must be recorded such as: fine sediment, cobble embeddedness, fines by depth, free matrix particle (Ries and Burns 1989), free winter particle (Espinosa 1991, 1992), quantity and quality of LWD (volume, length and diameter of pieces, presence of root wad, orientation, number of pieces), cross-section to bankfull (providing depth profile, bankfull width, bankfull depth, and channel unit volume), residual pool volume (Lisle 1987), V* or the fraction of the scoured pool volume filled by fine sediment (Lisle and Hilton 1991), water surface gradient (at base flow), and bank stability. For the monitored channel as a whole, overall channel gradient and water surface gradient should be measured as well as bankfull surface gradient. Hydrologic measurements (discharge) should be taken at established cross-sections during egg incubation and rearing periods for correlation with nearby gages to allow extrapolation of flow records to ungaged sites (see Orsborn 1990). Data must also be collected on riparian vegetation condition (tree density, basal area, height class, size diversity, vegetation types), condition of the riparian and floodplain surface (e.g., presence of roads, exposed surface, sediment source areas), sideslope condition, and water temperature.

Impairment of biological function of the Snake River basin from management-caused sedimentation is one of the most pervasive and serious threats to salmonid persistence. It is essential that sediment delivery, storage, and transport be controlled through effective watershed-wide planning based upon application of CSP land use standards, reliable data, and land management adjustments that are responsive to monitoring. As an indication of effectiveness of project activities (either development or restoration activities) in controlling sediment delivery, substrate sediment conditions and pool volumes must be measured. V*, or the fraction of scoured pool volume that is filled by fine sediment, provides a good index to sediment supply and mobility (Lisle and Hilton 1991). Trends in this variable can be monitored over a reach having no significant internal tributary entry points or sediment supply sources. Bedrock lithologic differences among watershed greatly influence substrate material composition and bedload transport rates via their control on hydrologic characteristics (runoff generation, stream power). Within this context, trends in sediment transport and sediment rating curves are affected by degree of development of a watershed.

Sediment delivery from all sources must be regulated to $\leq$20% over natural as assessed by the WATSED model for 4th-5th order watersheds after establishing a habitat baseline. Recovery trends in substrate sediment can be tracked in both Level 1 and 2 monitoring. Given the estimates of sediment delivery from Level 1 analysis, condition and trends in channel substrate plus in-channel stress-response from localized actions can be assessed at selected sites at Level 2. The RSI approach can be used to evaluate watershed stability and expression of surficial sedimentation and surface sediment mobility in intensively monitored channel types or channel units. Surficial sediment should be monitored with the transect technique (channel cross-sections, Platts et al. 1983). Sediment monitoring must include both estimates of surface fines and cobble embeddedness. In addition, there are several other kinds of monitoring related to sediment transport, storage, and substrate composition that would be advisable to include in a Level 2 monitoring program. These include measurement of free winter particle, free matrix particle, depth fines (shovel sample), suspended sediment, IGDO, intergravel velocity, bed scouring depth and volume (Nawa et al. 1988), and bedload.

Year-to-year tracking of habitat conditions along a salmon-bearing mainstem from one spawning or rearing area to the next provides a means to assess significant trends in the stream system, especially in terms of long-term recovery trajectories. This methodology may allow detection of localized impacts (e.g., on-site sediment impacts in tributary habitats) or propagating impacts along a river continuum (e.g., trend for increasing temperature or fine sediment) that can be useful for regression analysis with salmon or resident fish population data or community indices.

Fish populations should be monitored concurrently with the habitat system (watershed, riparian system, stream network, monitored reach). The frequency of habitat and fish population monitoring required depends on the degree of activity (logging, grazing, or recovery actions implemented) on the watershed. Spawning ground surveys have been a major emphasis of fish surveys in the Columbia River basin for at least 35 years. In the locations where surveys have been conducted over a significant time period, it is valuable to sustain this effort. Redd counts are generally made over index areas of at least 1 mile in length on an annual basis. It is recommended that index areas be held to a constant length at each site over the years of monitoring and that all counts be total (not simply peak) counts and converted to redds/mile. It is advisable to augment these counts with carcass surveys to assess male/female ratio and percentage of eggs expelled per female. It will be useful to establish Level 2 monitoring sites at selected key index areas and to assess from the Level 1 survey the characteristics of the index area in relation to the entire salmon-bearing portion of the stream. Juvenile growth studies would involve monitoring lengths, weights, and numbers on a weekly or bi-weekly basis. Macroinvertebrate community composition should be assessed for at least two seasons (e.g., early spring and late fall); invertebrate drift monitoring should coincide with the active summer growth season.

Spawning surveys should be conducted on the listed salmon species and steelhead, if flow conditions permit. Steelhead are generally present in greater numbers than salmon and occupy the same stream reaches. In addition, their distribution extends somewhat further upstream in tributaries, so they are useful indicators for upstream actions having negative impacts in salmon habitat. The spring spawning behavior of summer steelhead and the requirement for summer holding in pools prior to spawning of summer steelhead and spring chinook add extra capability to assess biotic response to seasonal habitat conditions. Water clarity should be recorded during periods of redd counts.

In addition to spawning surveys, assessment of juvenile densities of anadromous salmonids and juvenile and adult densities of resident salmonids and non-salmonids will be useful to fully establish the linkage between habitat condition and habitat recovery trends and fish species population densities, composition, and trends. Annual redd counts are a useful means of explaining trends in juvenile densities and to separate the effect of habitat condition from escapement strength on juvenile density. Because anadromous salmonids and certain other fishes are mobile after emergence, the juvenile densities measured in summer may not well reflect local spawning densities for the same brood year. For this reason, the Level 1 survey may be the best tool for estimating the percentage survival to summer or autumn from a watershed-wide egg deposition. However, the Level 2 survey should attempt to derive a comprehensive fish assessment (total enumeration for all species) for the monitoring reach. If this survey is conducted in the same season in repeated years, counts among years can be effectively compared. If annual habitat conditions are monitored, abundance by age class can be examined for linkages to environmental variables that may control survival at egg-to-smolt lifestages, (e.g., this enumeration may also allow identification of critical lifestage "bottlenecks" which may allow inference regarding habitat "bottlenecks" (limiting habitats). For example, a regression of some index of the water temperature regime or winter flow conditions preceding sampling with abundance of 1+ juveniles can be performed. A regression of 0+ juvenile abundance with summer/autumn low flows would provide useful insight to overall population trends.

Because salmon survival in passage and the ocean vary from year to year, adult spawning and seeding of the tributary habitat are variable. Consequently, this should be measured as accurately as possible by redd counts or weir counts for monitored tributaries to derive the greatest explanatory power. As support for interpretations of fish habitat trends made from anadromous fish surveys, Level 2 monitoring of resident salmonids is useful by virtue of eliminating most effects of downstream habitat- or management-related mortality (except in the case of resident fish that migrate extensively within a drainage basin). Because resident salmonids have generally similar habitat requirements to salmon, a decreasing trend in resident salmonid population abundance would imply a decline in habitat conditions that will affect salmon. In addition to resident salmonids, sculpins, for example, can be useful indicators of habitat with coldwater temperature regimes and clean substrates (Finger 1979) and can be counted by snorkeling.

Fish species composition provides considerable insight into habitat trends. Presence of exotic or resident non-salmonid species that reflect adverse habitat conditions for salmonids (e.g., warm water, abundant fine sediment, low dissolved oxygen) or severe competitive threat may provide a good explanation for low salmonid population size. Age (or size class) composition of salmonids serves as a good index of past overwinter habitat conditions and year-to-year habitat stability. Streams with poor quality overwinter habitat or a lack of overwinter habitat do not provide conditions suitable for overwinter survival. Under these conditions, 0+ age juveniles must migrate downstream in the autumn to larger river reaches or else they will suffer a high percentage mortality under winter flow conditions.

Approach

Identify channel types to monitor. Create detailed maps of the study reaches and stratify by channel units. Distinguish channel units used as spawning sites. Mapping by channel unit should include distribution of substrate types, LWD, channel width and depth, flow measurements at critical periods, associated riparian community and stand characteristics (height, cover, shade), bank stability. These data may provide a clearer relationship between watershed condition (or riparian condition) and in-channel habitat condition.

Compare data from replicates of fixed spawning or rearing sites among reaches to investigate the relationships between land use or condition and habitat condition within and among watersheds; between habitat condition and biotic indices for selected portions of streams incorporating spawning and rearing sites.

Evaluate trends among sites within a watersheds or trends among watersheds of a class subjected to preferred land management practices as a measure of effectiveness in restoring desired in-channel habitat conditions.

Compare population abundance trends among study sites of similar type to assess recovery in relation to habitat condition trends. Evaluate populations in several watersheds.

In 1/*j*th of a District's major salmon-producing streams select at least 4 spawning sites, 4 summer rearing sites, and 4 winter rearing sites for Level 2 analysis. All sites should be at least 100 m long. If possible, 1 or 2 of these streams should be in pristine drainages to provide some reflection of "control" conditions [note: see the Rhodes et al. (1994) "Range of Natural Variability" section for explanation of why such control sites may not found and, if they are found, why their condition may not necessarily indicate the long-term central tendency]. The test streams would be degraded to varying degrees but have restoration potential. Without availability of high quality, low management-impact watersheds, it is essential that Level 1 analysis provide an overview of watershed condition so that effects on selected habitat units or a series of such units on a river continuum can be interpreted effectively. That is, the physical or biologic behavior of any individual stream reach may be highly related to the spatial distribution of conditions immediately up- or downstream. In-channel conditions of a single reach may not reflect pre-management conditions even if the riparian zone remains pristine when the upper watershed has been managed. A Level 1 survey will provide needed data on overall watershed condition as well as spatial distribution of in-channel and riparian and watershed condition that could lead to improved ability to explain longitudinal shift in condition on a river continuum as a function of magnitude of impact and spatial proximity of upstream sources of impact to successive reaches on a continuum.

Selection of at least 4 sites of the types mentioned could be done by various alternative means, which lead to different kinds of insights on restoration. In addition to the contrasting of sites from pristine versus managed streams or watersheds, other sites to monitor or compare may be provided by certain opportunities or analysis needs. Recommended directions for selection of spawning areas include:

- long-term data set available. Select existing spawning areas having at least 10 years of redd count data. These sites may represent the best salmon spawning areas. Long-term biologic data provide important insights into habitat quality trends in a watershed and are even more useful when multiple data sets are available in a large watershed or subbasin. Sites with resident fish data in addition to salmon data are preferable.

- comparison of restoration trends in historic versus currently used habitats. Select a set of monitoring sites from a list of historic spawning grounds. These will provide a cross-section of low to high quality spawning sites. This will also provide the ability to examine the population and habitat trends for sites of different characteristics (e.g., different gradient, width, flow, temperature regime, drainage area, etc.). The importance of this monitoring design is that sites having the geomorphic potential to act as spawning reaches are evaluated for recovery trends. It does not restrict monitoring of recovery to only those remaining sites of high enough quality to function as spawning reaches.

- comparison of spawning (or rearing) data among similar spawn (or rearing) reaches based upon the characteristics of the site and environment. Select a set of spawning sites having similar physical characteristics by employing a stratification procedure. This will allow trends in quality of the spawning reaches to be examined relative to trends in their immediate riparian zones, the condition of the riparian zone for the stream network contributing to the spawning reach, the adjacent upslope contributing area to the spawning reach, and the condition of the watershed contributing to the spawning reach.

● examination of spawning or rearing data from similar or different types of reaches along a river continuum. Select a set of spawning reaches along a river continuum. These reaches will provide information on cumulative impacts propagating downstream in a primary watershed. The downstream changes in quality of spawning habitat can be related to progressive changes in watershed condition, stream potential in relation to increasing watershed area and fish species use.

Key issues to be addressed

a. In-channel and watershed trends.

1. What are trends in in-channel habitat conditions under various kinds and intensities of land management in selected channel units in 4th-5th order watersheds in defined-use areas (e.g., spawning, rearing, overwintering)? Examine aggregate effects of continued or decreasing stress levels on maintenance or restoration of habitat conditions.

Land use variables involved in monitoring:

All CSP land use parameters. This includes all variables used to estimate sediment delivery as percentage over natural; condition of the entire riparian network; sideslope condition.

Habitat variables involved in monitoring:

- Same CSP habitat parameters for which standards are set and for which no standards are set that are identified at Level 1.
- Trends in fine sediment at depth and surface fine sediment; bank stability.
- Trends in water temperature maxima, minima, and diel fluctuations in relation to trends in riparian network condition.
- Estimate cobble embeddedness in rearing areas to ±5% precision using techniques such as the ocular method, hoop method, and modified hoop method. ● Independently estimate percentage surface fine sediment in these areas.

2. What is the relationship between in-channel habitat condition in selected channel units of a salmon-bearing stream that meet defined-uses by salmon life stages (e.g., spawning, rearing, overwintering) *and local riparian, floodplain, or sideslope conditions?* Examine effects of decreasing stress levels on maintenance or restoration of habitat conditions.

Land use variables involved in monitoring:

Assess riparian condition within the study reach and 200 m upstream of the study reach to a higher precision level than used in Level 1 analysis. Focus on shade, cover type, species composition, density, and size classes. Correlate with Level 1 riparian analysis.

Habitat variables involved in monitoring:

Trends in water temperature maxima, minima, and diel fluctuations in relation to trends in condition of the immediate riparian environment. Also, relate water temperature to riparian condition for entire upstream riparian network.

- Substrate sediment condition
- Bank condition
- Pool volume and frequency

b. Biotic responses to in-channel habitat condition and trend. How are these responses linked to local riparian, floodplain, or sideslope condition; to general watershed condition?

1. What are trends in redd counts in key spawning areas?

Continue to assess redd counts per mile in established index areas or establish new index areas.
Habitat variables involved in monitoring:

- CSP variables set as numeric standards: percentage surface fine sediment
- CSP variables not set as numeric standards: water flow and timing, water temperature during spawning
- Classification: channel unit, drainage area upstream; riparian zone (late seral community, current community, cover, age, slope, soils); sideslope (percentage cover, current community, age); watershed conditions (sediment delivery, road density, percentage harvested, percentage in agriculture).

2. What is the survival rate of eggs deposited in spawning gravel?

Assess total redd count in the study reach; estimate total egg deposition; count 0+ age class as soon after emergence as possible to minimize reduction in population size from mortality early in the juvenile rearing period.

Habitat variables involved in monitoring:

- CSP variables: percentage surface fine sediment, water flow and timing during incubation.
- Additional monitoring variables that may be significant: depth of scour, gravel composition (particle distribution) to egg pocket depth, channel morphology (W/D, depth), egg pocket fine sediment; water temperature during egg deposition and adult female holding.

3. What is the population density of the 0+, 1+, and 2+ age class measured in late summer or early autumn rearing habitats and how does it relate to habitat conditions?

Adverse environmental conditions result in excessive mortality between year classes, resulting in a species population with a high proportion of young-of-year relative to older age classes. Year-to-year environmental variability, though, can contribute to significant annual variation in the ratio of 0+ to older age classes (Austen et al. 1994). Environmental trends

also can lead to periodic or long-term, progressive increase or decrease in available rearing area.
Estimate numbers by size class using snorkel diving. Confirm age for size classes by scale analysis.

Habitat variables involved in monitoring:

- CSP variables: cobble embeddedness, LWD, bank stability, water temperature (maximum summer, minimum winter), water quality (turbidity), streamflow discharge (peak flow magnitude and frequency, summer baseflow).
- Additional monitoring variables that may be significant: depth and velocity contours of channel unit, presence and abundance of competitor and predator species.

Classification: channel unit distribution in channel types; proximity to higher quality areas.

4. What are the overwintering densities of juvenile fish?

Estimate numbers of overwintering juveniles by size class by snorkel diving during winter; numbers of late summer or autumn parr available prior to winter period.

Develop a better understanding about what constitutes effective overwinter habitat.

Overwinter survival is a function of environmental factors such as flow or temperature extremes and physical habitat quality (e.g., availability of deep pools, abundant LWD, undercut banks, cobble and boulder interstices free of sediment, etc.). Overwinter survival should be monitored by channel unit in Level 2 and integrated at the spatial scale of a logical salmon production unit (e.g., 4th-5th order) in Level 3.

Habitat variables involved in monitoring:

- CSP variables: cobble embeddedness, LWD, bank stability, water temperature (minimum winter), water quality, streamflow discharge, pool depth and frequency.
- Additional monitoring variables that may be significant: depth and velocity contours of channel unit, bank overhang, distance from bank, substrate large particle size distribution, precipitation as snow, snow depth, channel icing.

5. Assess the magnitude of autumn migration of juvenile salmon.

Using smolt traps or weirs and traps, count numbers of juveniles outmigrating in autumn relative to numbers remaining in the reach.

Reaches without suitable winter habitat cause fish to seek cover in autumn. This likely causes overwinter mortality that is in addition to that which would have resulted from adequate wintering habitat being available in all reaches.

Habitat variables involved in monitoring: same as in 4 above.

6. Biotic indices. What do biotic indices reveal about trends in habitat quality or progress in restoring the watershed?

i. What are the trends in abundance and composition of the resident native fish community?

Resident coldwater stenotherms should be monitored as indicators of probable effects on anadromous salmonids (especially the listed species). Determine population density and size class distribution by channel type and channel unit. Among those fish recommended as surrogates for anadromous salmonids, which may or may not be present in sufficient numbers for monitoring, are several sculpin species, bull trout, cutthroat trout, and rainbow trout. Many sculpins and bull trout are sensitive to stream temperature warming, fine sediment, and LWD loss. Frissell (1992) suggests use of cutthroat trout as a sensitive indicator of thermal increase in place of steelhead/rainbow.

Habitat variables involved in monitoring:

- CSP variables: surface fine sediment, cobble embeddedness, LWD, bank stability, water temperature, water quality, streamflow discharge.
- Additional monitoring variables that may be significant: depth and velocity contours of channel unit

ii. What are the trends in abundance and composition of the resident exotic fish community?

Distribution, density, and age (or size) classes of exotic fish should be monitored by reach classes (channel type, channel unit) stratified by ecoregion and watershed class and also by major management impacts (e.g., reaches in allotments, heavily roaded stream reaches, reaches that were clearcut or are undergoing various management treatments). Numerous exotic fish (non-native) contribute to the displacement of native fishes, some of which are listed or are state sensitive species. Many of these exotic fish are tolerant of degraded conditions and the upstream extent of their distribution or their population strength in various reach types indicates the trend in their potential threat to indigenous fishes.

iii. What are the trends in abundance and composition of the macroinvertebrate community?

Quantitative estimates of total density and biomass rather than relative estimates; resolution to taxonomic species level rather than genus, family, order, or functional feeding groups; includes identification of chironomid species. Otherwise, see criteria in Level 1.
Measure for selected channel units within selected channel types.

iv. What are the trends in abundance and composition of the riparian community?

Measure with respect to channel types, geomorphic surfaces, and existing discontinuities in the riparian community.

v. What are the trends in vegetation cover on transects perpendicular to the stream channel bank and outward to a distance of 300 m?

Use remote sensing satellite imagery to detect effects of grazing on range and riparian vegetation. Vegetation parameters that are useful indices to vegetation condition, cover density, and resilience to a combination of rainfall and grazing events include wet period average cover, wet period cover variance, vegetation response to rainfall, and rate of cover depletion (Pickup et al. 1994).

Level 2. Intensive Trend Evaluation: Non-Salmon

Geographic Unit for Analysis

Non-salmon-bearing headwater tributary to secondary tributary level. Example progression of streams: Mex Creek to White Sand Creek

General description

Intensive monitoring of selected portions of a set of headwater tributaries provides the ability to determine the response (or range of responses) to watershed (or riparian) management stresses or recovery actions or trends. Trends of replicate stream sites in response to specific nearby riparian, floodplain, or sideslope conditions or to the general state of the upstream watershed constitute effectiveness monitoring of practices designed to maintain stream channel condition or improve condition. This level of monitoring is not as scientifically rigorous as in Level 3 because many extraneous variables are not measured or controlled, and their effects may not be removed from main effects (e.g., climatic shifts). However, explanatory power is improved by virtue of the number of similar reaches that are evaluated, the number of years of trend data demonstrating the effectiveness of the action, or the ability to identify specific sources of local impact and initiation time of impact, and also efficiency of the classification systems to distinguish streams, reaches, or smaller portions of reaches (e.g., units of habitat).

Objectives

The objectives for Level 2 monitoring of non-salmon-bearing watersheds include determination of response of specific channel types or channel units to a known, local impact or to an estimated level of cumulative impact in 2nd-3rd order watersheds. The response provides an estimate of effectiveness of management actions in moderating impacts relative to other methods or in producing recovery. Response can be measured as a change in key in-channel habitat parameters such as fine sediment storage, heat load, LWD storage and distribution, residual pool depth). The measured response can be a deviation from standards or from baseline conditions, or trends in similar channel units upstream and downstream from point of impact. Monitoring of response to impacts, and especially to change in input functions, can be used to assess effectiveness of land management. Level 2 monitoring of low order tributaries to salmon-bearing reaches is intended to record "early warning" signals to eventual downstream cumulative effects.

Level 2 monitoring of non-salmon-bearing watersheds should be used to provide information that can be used to document effectiveness of BMPs on a regional basis. Effectiveness of practices within a region should be tabulated for an array of environmental conditions (e.g., climatic zone, underlying rock or soil type, slope gradient, vegetation zone, etc.) for specific reproducible land management actions (e.g., construction of a specific type of road surface, a specific grazing system). An effectiveness monitoring catalogue provides a structured basis for recommending site-specific actions and identifying new ones whose effectiveness must be assessed. This information is critical to the success in applying the R1/R4 sediment model to provide sediment yield estimates approximating actual absolute values.

Progress in completing the BMP catalogue (effectiveness of actions under various environmental conditions or regimes) should be reported annually. Actions that are taken whose effectiveness is not known should be reported. The limits to known effectiveness should be explained in relation to expected environmental regimes. That is, if slash-filter windrows are effective in preventing 99% of sediment from reaching the stream channel under 5-year storm events but may fail completely in a 10-year event, these limits to knowledge and the risk involved should be thoroughly detailed. This kind of analysis could provide error limits to predicted sediment yields. Finally, land managers should provide a
plan for eliminating the information gaps regarding effectiveness of actions in producing a desired effect.

Level 2 monitoring in non-salmon bearing tributaries should occur annually in a fixed set of tributaries to reflect long-term trends. If possible, an equal number of fixed, unmanaged control sites should be selected. However, for specific BMP effectiveness evaluation or project monitoring, it may be necessary to introduce additional test streams whose watersheds are receiving a management action. For this reason, it is important to establish enough control sites in minimally perturbed tributaries so that major stream types (e.g., varying by characteristics such as geology, gradient, elevation, etc.) can be represented in long-term trend data sets defining BMP effectiveness, recovery rates, response times, input, storage, and transfer rates, and biotic community characteristics and dynamics. This monitoring should emphasize measurement of discharge, sediment yield and delivery, percentage surface fines, sediment storage volume behind obstructions, residual pool volumes, riparian condition, and temperature regime. The current capability of a stream to transport fine sediment can be monitored using tracer sediments (Ketcheson and Megahan 1991). The amount of fine sediment retained in reaches tends to be a function of channel gradient, effective gradient, channel sinuosity, flows, channel cross-sections, sediment supply, bedforms, LWD volume, and floodplain characteristics.

Approach

Focus effort on monitoring representative, sensitive, or strategic channel types or channel units as response units to management actions in adjacent riparian or watershed areas or to a single upstream activity or combined set of upstream activities. This monitoring acts as an early warning of negative changes that are likely to portend long-term, slowly reversible effects.

Monitor long-term trends in in-channel habitat riparian, and watershed condition and in biotic responses as evidence of effectiveness in achieving restoration.

Channel type or channel unit performance will be measured typically by either evaluating a habitat parameter for the unit in totality (e.g., estimation of all LWD in the unit; ocular estimation of percentage of surface sand), by taking random samples within the unit, or by taking samples on transects that are spaced at regular intervals longitudinally.

Key issues to be addressed

a. In-channel habitat condition

1. Determine in-channel effectiveness of riparian restoration in improving the temperature regime on a set of stream reaches. Conversely, if riparian impacts are planned, monitor the shifts in temperature regime, sediment regime, channel morphology, and LWD.

Environmental conditions involved in monitoring:

- CSP variables: water temperature (maximum and mean daily temperatures), fine sediment, residual pool volume.
- Additional variables: stream channel width, reach length, reach orientation, riparian community characteristics (dominant species, age, height, percentage shading), water discharge, monthly mean water depth, groundwater entry.

Classification: channel type, riparian community type.

2. Estimate the sediment loading rate in storage sites (behind boulders, upstream of LWD, channel bars, pools, floodplains) of 2nd-3rd order channels. Estimate the LWD loading rates concurrently.

Environmental conditions involved in monitoring:

- CSP variables: percentage surface fine sediment, cobble embeddedness, LWD (volume, number, diameter classes), pool frequency and volume, residual pool volume
- Additional variables: total sediment storage, sediment particle size composition and d_g, water discharge regime (peak flows), volume of sediment associated with a low gradient reach above a control point as measured with regularly spaced channel bed surface cross-sectional profiles.

Classification: channel type, channel unit type

3. Estimate the rate of sediment and LWD delivery to, storage in, and transfer from zero-order and first-order channels.

Environmental conditions involved in monitoring:

- CSP variables: estimate surface fine sediment
- Additional variables: total sediment volume delivered and in storage, LWD loading (volume, number, diameter classes), water discharge regime (peak flows), occurrence of mass failures; riparian community characteristics (stand density, age diversity, species composition).

Classification: landtype, sideslope gradient, geologic rock type.

4. Monitor trends in residual pool volume, pool frequency, and total pool volume.

Environmental conditions involved in monitoring:

- CSP variables: LWD (volume, number, diameter classes), pool frequency and volume, residual pool volume, total pool volume, bank stability.
- Additional variables: water discharge regime (peak flows), volume of mass failures (sideslope failure), sediment transport into the study reach from upstream watershed, sediment transport from tributary reaches entering the study reach; sediment delivery estimated from Level 1.

Classification: channel type, W/D, sinuosity, bed material size, stream power.

5. Monitor trends in the temperature profile measured longitudinally along the tributary. The downstream temperature profile, accompanied by a map of groundwater and tributary water entry points (volume and temperature of water additions), provides a broad view to the state of the riparian system and alerts one to problem locations where abrupt deviations occur or greater than average rates of stream warming occur. Progressive shifts in the annual downstream trend of maximum daily temperature can indicate general or local recovery or degradation of the riparian zone or watershed condition.

Environmental conditions involved in monitoring:

- CSP variables: water temperature (maximum daily temperature), riparian shading, residual pool volume, channel width, stream flow discharge.
- Additional variables: maximum daily air temperature at known elevations, topographic relief, daily precipitation during summer, mean albedo over watershed area and cloud cover, surface water temperatures as measured by aerial thermography.

b. Biotic responses

1. Monitor the rates of riparian recovery and trends in in-channel habitat conditions in grazing exclosures (excluding all ungulates) relative to riparian areas without exclosures.

Environmental conditions involved in monitoring:

- CSP variables: bank stability, percentage surface fine sediment, pool frequency and volume, residual pool volume, stream shading.
- Additional variables: dominant riparian vegetation (age, species, height, shading), groundcover (dominant species, cover), shrub cover (dominant species, cover).

Classification: channel type, riparian community type, riparian landform

2. What are the biotic responses to in-channel habitat condition and trend that may be attributed to local riparian, floodplain, or sideslope condition; to general watershed condition.

Environmental conditions in monitoring:

- CSP variables: percentage surface fine sediment, fines by depth, bank stability, water temperature, cobble embeddedness.
- Biotic variables: density and age (or size) composition of bull trout, cutthroat trout, sculpins .

Classification: channel type, channel unit, riparian community type, watershed class and condition.

Level 3. Validation of Salmon Population and Salmon Habitat Response

Geographic Unit for Analysis

Salmon-bearing watersheds from the scale of secondary to primary tributary to large watershed sub-basin level. Example progression of streams: Fish Creek and White Sand Creek, Lochsa Creek, Clearwater River, Snake River.

General description

Intensive or extensive, validation monitoring focused on salmon survival and production from the entire stream system; also validation of models at the reach, network, or watershed scales linking land management with in-channel habitat and biotic response. In addition, this validation monitoring allows calibration of monitoring methods used in lower intensity, broader geographic coverage.

Develop and validate methods linking salmon survival and production to habitat condition.

Develop models of fish community composition as a function of habitat quality, stability, and spatial organization. Validation should be relative to watershed, riparian, floodplain, and hillslope class (or landtype).

Objectives

Level 3 monitoring for salmon-bearing watersheds is to validate assumptions forming the framework of the land management system, including efficacy of its standards, to calibrate methods across levels of analysis (e.g., Level 1, 2, and 3), and to assess the bottom-line assumption in managing salmon-bearing watersheds, that smolt output per spawner will increase in response to watershed restoration actions. Key assumptions of land management are that salmon survival and the capability to produce salmon smolts will not be reduced and will be increased through recovery allowed by improved land management.

Validation monitoring is monitoring of conditions or trends that can be used to improve or adjust model predictions or a set of assumptions about likely outcomes from employing a single land management procedure or set of procedures (e.g., evaluation of overall benefit of combined restoration actions). Validation monitoring of processes with rapid responses to a limited set of controlling variables are highly conducive to experimental manipulation (e.g., improvement of temperature model components that determine the influence of factors such as canopy cover, wind speed, boulders protruding from the water surface, etc.). Validation monitoring is typically construed as research by federal land managers, although this concept has often resulted in lack of progress in developing, improving, and validating important models used in land management. The explanation for this dilemma is that development of these models is often not perceived as research by the federal research stations, and the land management districts consider it to require too high a level of commitment, expertise, or funding to justify. Consequently, much monitoring falling on the continuum from effectiveness to validation is not conducted by anyone. Sediment, temperature, and LWD recruitment models for use at site (reach, hillslope) or watershed scales are very important tools needed to regulate the frequency and intensity of impacts of management actions (or recovery actions) to stream channels.

Validation monitoring intergrades with effectiveness monitoring because both types of monitoring can make use of data on in-channel or riparian condition and trend, and response to changes in condition or output from terrestrial land units. Although the interrelatedness of these types of monitoring makes it unclear whether the responsibility is primarily a management or research one, it underscores their necessity because the need for this feedback is acute for land managers. As an example of the continuum of monitoring objectives and ability to interpret monitoring results, if it is assumed that timber harvest or road construction at a district using a PACFISH buffer will not increase sediment delivery or downstream surface fines, effectiveness monitoring provides a check on the veracity of the assumption; if the data are carefully collected it can be used to calibrate or refine these assumptions and models (i.e., validation monitoring). Proactive validation monitoring of riparian function might consist of evaluating predictions of a temperature or LWD loading model for two or more decades after riparian revegetation commences.

The biological imperative, though, is for the riparian stand to provide needed levels of LWD, channel structure, desirable W/D, and temperature regime. It is difficult for one model to incorporate all these functions. In addition, the connection between shading and temperature is complicated by confounding factors so that, without careful analysis, it is not easy to demonstrate effectiveness. For example, because entry of cold groundwater to a particular reach can reduce water temperature from upstream to downstream points of measurement, removal of shade in this reach could be interpreted as producing no increase in temperature over this distance provided effectiveness monitoring was done as post-activity upstream-downstream monitoring rather than pre- and post-activity and the influence of groundwater was overlooked. Also, if effectiveness monitoring is done by measuring temperature on one day during a mild summer, the effectiveness of the riparian management program in controlling water temperature increases would likely be overestimated.

a. Validation of habitat response

Validation monitoring should evaluate predictions that watershed sediment delivery and fine sediment levels will not increase if low risk logging is planned in conjunction with effective road obliteration. Road obliteration, in this case, would be estimated to reduce sediment delivery by a magnitude much greater than the increase caused by logging itself. Such a prediction can be evaluated by validation monitoring in salmon-bearing and non-salmon bearing streams by measuring sediment delivery, sediment storage, surface fine sediment, and cobble embeddedness.

Validation and effectiveness monitoring are most efficiently conducted by employing stratification techniques (classification) to narrow the sources of variation in monitoring sites. This is accomplished by carefully defining the environmental system for the site being evaluated. Characteristics of the site and its environment (e.g., the reach and its riparian zone, valley, or watershed) set the boundaries under which the BMP or model is evaluated and operates. Behavior of the system might be explainable with respect to these characteristics. Similar behavior from similar sites under similar environmental stimuli helps confirm model predictions, when the model incorporates capacity and performance characteristics of the site and its environment explicitly (e.g., a particular landtype is covered by its potential natural vegetation in one site and is clearcut in another) or if the stratification is conducted as a precursor to application of the model. Examining reaches of different type or in different environments could obviously yield different results under the same perturbation (climatic event or management practice). In order to protect the most sensitive landforms or stream channels, it will be important to adequately classify these land or stream units so that responses can be evaluated from similar systems operating.

A key management assumption is that if a certain set of land management standards is employed at the watershed, riparian, and channel levels, as proposed in the CSP (Rhodes et al. 1994), significant and continuing progress will be made in restoring salmon habitat. Improvement trends using CSP recommendations can be compared with trends under approaches of PACFISH and FEMAT. In the CSP, the assumption of improvement is based on the ecosystem perspective that excellent quality fish habitat is produced from the interactive performances of a healthy watershed system, whose functions most nearly reflect those of the watershed in its native, unmanaged state. This state is approached to some degree in the effort to constrain activities. One significant constraint on level of activity is accomplished by applying a sediment delivery model so that mean annual total sediment delivery (as a percentage over natural) is regulated, by adjustment of level of management impact on various landtypes, to a value that is low enough to be within transport capacity of the channel and may allow channel recovery. Rhodes et al. (1994) considered that 20% over natural sediment delivery is a minimum starting point from which to begin to expect monitoring data to show an improving trend. To the extent that natural functions can be restored to all components of the watershed system, the aquatic system will benefit.

To validate this assumption is not a simple matter. The complex interactions among all components just identified in the watershed system form the basis for decades of research in watershed science. However, the management standards proposed in the CSP emphasize the predominating influence of a high quality riparian zone on the stream system and the importance of managing the entire watershed to control sediment delivery to the stream system. Implementation of a system of riparian reserves extending to the smallest headwater tributaries and strictly controlled management of sediment delivery (among other standards proposed in the CSP) provide a form of holistic management for which benefits can be validated through monitoring. Actually, these assumptions can be validated and linkages established between land use and in-channel habitat condition through a combination of Level 1, 2, and 3 monitoring. The extensive Level 1 monitoring provides trend data for entire stream systems (of different types) that start from different degrees of degradation but may all be influenced by the same management standards in the future (i.e., if CSP land use standards are uniformly applied). Progress in recovery can be assessed from a broad scale (Level 1) look at habitat and population trends for salmon- and non-salmon-bearing streams or as a watershed-wide recovery in riparian vegetation. Spatial distribution (mapped by GIS) of riparian vegetation characteristics may be investigated for discernible relationships to distribution of bank condition, longitudinal temperature trends, fish species or community distribution, or limits of spawning and rearing. Level 2 monitoring focuses on response of specific in-channel habitat characteristics and more intensive fish population estimates (trends in listed species, species diversity) at sites selected for strategic or representative purposes. In Level 2 analysis, the effect of trends in local riparian condition and network riparian condition can be investigated for their effects on trends in intensively monitored reach habitat characteristics (e.g., LWD, water temperature, substrate characteristics). Model validation will result from Level 3 analysis. An example of such validation monitoring is comparison of quantitative predictions made from a water temperature model of longitudinal temperature profile based upon trends in canopy cover in riparian reserves. Temperature at any given point can be modeled in relation to riparian cover expressed at various spatial scales (e.g., local reach versus total upstream riparian condition effects). A model of sediment delivery must be linked to substrate condition.

For sediment delivery/substrate condition validation, riffle and glide areas should be monitored for fines in the bed by depth (McNeil core using at least a 12 inch diameter core, dry sieving) as well as surface substrate composition and embeddedness to derive correlations among the methods. Auxiliary methods are free matrix particle (sampled concurrently with embeddedness) during summer/autumn, and free winter particle (Espinosa 1991, 1992) established on a transect through winter rearing habitat.

In the same locations sampled for sediment storage and all fine sediment measures, establish scour monitoring devices (Nawa et al. 1988) along selected transects across major bars and riffles or spawning areas (for salmon-bearing reaches). The volume of sediment in transition on the bed during various size storms will provide information on bed stability and can be linked with the RSI and bedload estimates. In spawning reaches, measures of depth and spatial distribution of scour provide information relevant to ability of incubating eggs to survive at different points down the mainstem.

Level 3 monitoring conducted in a salmon-bearing stream should be performed in at least one of the streams in which Level 2 monitoring is done to provide important correlations among scales of analysis, intensity of sampling, and precision of methods employed.

b. Validation of biotic response

Level 3 habitat monitoring is comprised by validation of various models relating land use or condition of the watershed or riparian zone to in-channel habitat conditions. However, biotic response is the bottom line. Population monitoring under Level 3 emphasizes fish production on a watershed scale, incorporating measures of adults in, survival of eggs and juveniles during rearing, and smolts out. On a regional basis there are very few streams in which smolt production has been related to adult spawning and summer rearing and overwinter habitat quality. From the viewpoint of management of freshwater stream systems and their watersheds, not involving passage concerns, smolt production (quantity and quality of smolts, and timing of outmigration) and survival are of major importance in population abundance.

To conduct validation monitoring of a salmon production unit, the following procedures are recommended: Select a salmon-bearing stream having a variety of land use practices in the watershed. The watershed should be one that is also monitored with Level 2 procedures and is large enough to be able to provide habitats for juveniles throughout the rearing stages (including overwintering) currently or if conditions improve so that these habitats are restored or reconnected with other portions of the watershed. Establish a weir for monitoring adult immigration to the watershed. Using video technology record daily the number of adults that migrate upstream through video counting passages. This technique allows determination of species, size, and numbers. Determine the distribution of spawners throughout the watershed. Record data as a GIS data layer and analyze spawner distribution in relation to channel type, discharge at the site, daily temperatures (maximum, minimum), proximity to deep pools, LWD, bank cover, and riparian canopy, and quality and quantity of substrate in spawning areas of the channel.

Record the locations for adult holding prior to spawning. Identify the volume and size of the pools and associated habitat characteristics. Determine whether the pool provides a coldwater refuge owing to depth, subsurface seepage, or entry upstream of coldwater tributaries. Determine numbers of adults holding in representative pools (by depth, volume, area, location in stream system, associated pool cover features). Assess incidence of adult diseases (obvious visual signs) and pre-spawning mortality. Assess shifts in adult holding distribution during the pre-spawning season as water temperature changes.

Record the time of initiation and termination of spawning behavior for major spawning areas. Assess differences by position along the mainstem and in tributaries in relation to water temperatures. Determine age, size, and sex of adults from carcass surveys.

Monitor changes in fry density during the emergence period in selected spawning areas to determine timing of emergence. Relate emergence to water temperature during the emergence period and cumulative degree-days from the period of peak spawning. Census fry densities in relation to redd counts by spawning area as an estimate of egg-to-fry survival. Continue to monitor the lower limits of juvenile distribution in the mainstem and tributaries during the summer/autumn rearing season to assess the influence of water temperature and streamflow on in-basin migration. Relate distribution of juveniles to channel unit type; to presence of cover factors within channel units (water depth, LWD, overhanging banks), or positions within channel units (e.g., availability of complex margins). Investigate the suitability of new means to estimate in-channel cover, such as the method of Kinsolving and Bain (1990) that incorporates cover density, complexity, and spatial heterogeneity. Estimate total basin autumn densities to assess significance of mortality factors existing during the spring/summer rearing period following emergence. Assess size and condition of fish entering the winter period. Calculate growth rates for the period from peak emergence to the autumn sampling period.

Determine the winter distribution of juveniles. Record the types of instream characteristics that provide good overwinter habitat. Assess relative number of overwintering juveniles in these various habitat types, water temperatures, presence of ice, annual flow peaks. As soon as flows permit, map the distributions of all probable overwintering sites as a GIS data layer based on extrapolation from winter observations. Snorkel representative sites by day and night to estimate densities. Assess the importance of off-channel ponds and wetlands as overwinter rearing sites relative to the main channel.

Estimate overwinter survival from the ratio of total basin smolt counts to counts made in the previous autumn.

Assess annual trends in total basin smolt output. Determine smolts produced in relation to potentially spawning females, and/or to redd counts, and estimated number of deposited eggs.

Assess the abundance and diversity of drifting macroinvertebrates in summer rearing sites along the course of the mainstem. Determine abundance in relation to volume of water filtered and the flow rates across rearing areas. This measure is an index to food availability and food quality for salmon juveniles.

Implant pit tags into trapped, naturally produced smolts to assess survival in downstream migration through dam counting stations. Implant coded wire tags to determine smolt-to-adult survival for a group of smolts. Retrieve tag codes, scales, size, and sex information from carcasses of adults containing pit tags.

Assess smolt trap efficiency from smolt trapped fish that had pit tags implanted and were released above the trap. Assess differential day versus night trap efficiency because it has been found that visibility of the trap results in avoidance behavior.

Determine annual trends in biotic factors listed above as habitat recovery occurs or populations recover to fully occupy available habitat that currently exists. Compare the trend to those of other streams in the region having Level 3 monitoring. Extrapolate results to other streams that have had only Level 2 monitoring.

Approach

Consider the study unit to be the entire secondary tributary stream system as a production unit.

Establish a weir on important salmon-producing tributaries for enumeration of adult immigration and smolt emigration.

Comparison of fish production from one stream system to another will be made on the basis of habitat quantity and quality, watershed type, and condition.

Key issues to be addressed

a. In-channel habitat conditions

1. Document trends in overall habitat quantity and quality associated with application of land use standards and in-channel habitat standards as a means to initiate management feedback. Develop a model of salmon carrying capacity to provide estimates of biologic value of basin-wide and reach-scale improvements in habitat quality. Estimate degree of increase in quantity of available habitat (e.g., increase in downstream extent of coldwater zone, increased volume of habitat due to improvement in pool volume, bank overhang, LWD volume, reduced embeddedness, etc.)

2. Measure depth of scour and deposition in key spawning areas. Correlate with streamflow statistics, RSI, q* (measure of difference between surface and subsurface substrate composition. Estimate spatial extent of scour. Estimate potential impact on survival of incubating eggs of salmon species.

3. Evaluate the correlation between CSP sediment indices and other sediment indices (percentage surface fine sediment, depth fines, free matrix particle, free winter particle, visual estimation of surface particle size frequency, geometric mean particle diameter). Evaluate the distribution of sediment particle sizes in spawning gravel to assess the degree to which surface fine sediment would represent the potential salmonid response (i.e., the degree to which coarse particles or skewed particle distributions could alter the negative effect of fine sediment on egg survival).

4. Validate the USFWS reach temperature model in the Snake River Basin tributaries. Compare its performance with another reach level model (e.g., Beschta and Weatherred 1984). Use these models to predict water temperature trends in relation to trends in riparian community condition.

5. Evaluate LWD loading models to assess projected rates of recovery of LWD, given current riparian types and condition, under a passive recovery scenario.

6. Confirm that bank stability and W/D will improve by reduction in sediment delivery and application of the riparian reserve system. Evaluate the sensitivity of riparian landforms to these watershed scale and riparian management alterations.

b. Watershed habitat conditions

1. Calibrate the variant of the R1-R4 sediment delivery model used in the particular region. Determine appropriate background sediment delivery for various landtypes and drainage densities. Develop and/or improve methods for estimating sediment delivery from mining, grazing, and tilled agriculture. To date, the sediment model has not been well documented for use on large watersheds. Measure sediment delivery in relation to the combination of drainage density and road density. Road density is cited as a principal factor related to increasing sediment delivery to the stream channel network. Watersheds with lower road density, and especially road density in the riparian reserve, should have lower sediment delivery rates. Estimate sediment delivery for several tributary watersheds. Validate the estimation procedure.

Environmental conditions involved in monitoring:

- CSP variables used in estimation: road density, road age, drainage density, percentage of watershed clearcut within last 10 years, percentage of watershed clearcut within last 25 years. These variables were monitored in Level 1 analysis.
- CSP variables used in validation: percentage surface fine sediment, cobble embeddedness, LWD (volume, number, diameter classes), pool frequency and volume, residual pool volume. Data on these variables were collected in Level 1 and 2 analysis.
- Additional variables: total sediment storage, sediment particle size composition (d_g), water discharge regime (peak flows).

Classification: watershed type (drainage density, relief).

2. Validate the sediment delivery/substrate response model (USFS 1983) prediction of long-term substrate recovery by evaluating current percentage surface fines and cobble embeddedness and trends; determine whether trends in substrate indices can be predicted from trends in estimated sediment delivery; investigate effects of short-term climatic events on sensitive portions of the landscape to substrate sediment trends.

3. Calibrate the RSI method relating watershed development to riffle stability under the environmental conditions presented in the Snake River Basin.

4. Assess the sensitivity of variables other than sediment indices that are affected by increasing/decreasing sediment delivery such as residual pool volume, pool frequency, total pool volume, and stored sediment volume.

5. Derive the relationship between bedload transport and flow for a salmon-bearing watershed of mapped inherent characteristics (e.g., bedrock type, drainage density, relief) and state of development. The state of development is a surrogate for mean sediment delivery, which establishes the sediment supply to which channel morphology and bedload transport adjust.

6. Measure the total sediment budget (sediment delivery, transport as bedload and suspended load, storage) for a salmon-bearing watershed. Document inherent characteristics (e.g., bedrock type, drainage density, relief) and spatial states of land management impacts.

7. Evaluate the use of watershed structural indices (e.g., drainage area, relief, drainage density), watershed characteristics (soil depth, rock type, etc.), and climatic regions to predict peak flow and low flow statistics among watershed types. Such analyses should allow hydrologic regionalization, which may provide a physically based framework for conducting monitoring of populations or communities.

c. Biotic response

1. Determine fish survival rates.

i.. Adult migration into secondary tributaries and survival in holding areas. Use video counting technology for counting fish passing a temporary weir. Make snorkel counts of adults in holding areas.

Habitat variables involved in monitoring: primary pool frequency, area, and volume; percentage cover within pools.

ii. Assess total egg deposition. Count adult spawners and total redds; determine fecundity; assess mean percentage of egg retention by females; assess male/female sex ratio; size and age frequency of spawners.

iii. Egg-to-fry survival. Assess fry numbers periodically after emergence using Hankin and Reeves method.

Habitat variables involved in monitoring: percentage surface fine sediment, fine sediment at depth, fine sediment in egg pocket, water quantity and timing, area of spawning habitat, temperature, overwinter sedimentation.

iv. Fry-to-pre-smolt survival. Assess 0+, 1+, and 2+ age class abundance using Hankin and Reeves method in spring and late summer or early fall and attempt to relate this to habitat conditions.

Habitat variables involved in monitoring: cobble embeddedness, large woody debris, primary pool frequency and volume, residual pool volume, bank stability, water temperature, stream shading, water quantity and timing, macroinvertebrate community composition and abundance, water quality, fish community composition and abundance (predator, competitor abundance).

v. Pre-smolt-to-smolt survival. Assess smolt output using smolt trap and investigate relationships to habitat conditions. Use pit tagged smolts to assess sampling efficiency.

Habitat variables involved in monitoring: primary pool frequency, area and volume; cobble embeddedness, large woody debris, bank stability.

vi. *Smolt survival during migration.* Measure by use of pit tags from secondary tributary to Lower Granite Dam and investigate relationship to habitat conditions.

Habitat variables involved in monitoring: water temperature, water timing, water travel time, fish community composition (exotics, predator densities).

2. Determine seasonal fish migration patterns as related to habitat variables.

Habitat variables involved in monitoring: cobble embeddedness, large woody debris, pool frequency and volume, residual pool volume, bank stability, water temperature, water quantity and timing, fish community composition and abundance.

3. Determine fish population production from a secondary tributary. Measure abundance and biomass of fish at intervals during their growth period. Fish growth rates indicate suitability of their environment bioenergetically to produce high quality smolts (i.e., sufficient size at emigration to achieve high survival rate on ocean entry; lack of stress factors leading to poor condition factor or disease).

Habitat variables involved in monitoring: all variables listed for a(1-4).

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4. Evaluate the use of resident fish populations (sculpins, bull trout, cutthroat trout) as indicators of temperature and sediment trends. Correlate annual population size, community composition, and size class variation with environmental variation.

5. Establish an macroinvertebrate reference collection for streams of different types and condition. Assess the ability to define sediment-sensitive indicators. Relate community structure to habitat stability.

Level 3. Intensive Validation of Models and Assumptions for Physical Processes and Biotic Response

Geographic Unit for Analysis

Non-salmon-bearing headwater tributary to secondary tributary level. Example progression of streams: Mex Creek to White Sand Creek

General description

Validation monitoring focused on the key assumptions and models on which the land management approaches are based (e.g., CSP, PACFISH, etc.) are based plus those models incorporating CSP parameters that would improve the ability to estimate impacts from land management actions or degree of benefit from restorative actions.

Objectives

Level 3 monitoring for non-salmon-bearing watersheds should provide data for calibration and validation of models linking land management activities to in-channel habitat response or habitat quality, quantity, or spatial distribution to biotic response. Data for analysis can be drawn from condition and trend monitoring conducted in Level 1 and 2 analyses or from monitoring of habitat or biotic response in Level 2 effectiveness monitoring. Validation monitoring can be a matter of confirming a black box model in which, for example, predictions of improving habitat condition are related to the combined effects of numerous conditions in the local or upstream riparian zone or watershed. Otherwise, validation may investigate the sensitivity of certain model parameters in predicting habitat condition or trend. In this case, an accurate spatial delineation of watershed, riparian, or in-channel condition may be needed. For model validation at a site, there may need to be assurance that uncontrolled variables not included in the model will not bias results. If replicates are needed to represent varying levels of environmental control on a reach, classification techniques may be needed to compare condition and trend from similar systems (i.e., having similar sets of interactive, functional components and environments) at any geographic scale. Level 3 monitoring provides the best information base for conceptually linking habitat condition and response to land management because it expresses mechanisms quantitatively; has higher resolution due to smaller area (reduces overlapping signals from different parts of larger watersheds); has greater homogeneity of watershed environmental factors (e.g., precipitation, elevation); reduces dilution effects; and at these scales, unperturbed reaches in small subwatersheds can be found to act as monitoring controls for comparison with perturbed reaches.

A key model validation and model improvement challenge at Level 3 involves the WATSED sediment model. Land use development in headwater tributaries contributes fine sediment to lower salmon-bearing reaches of a mainstem secondary or primary tributary. Control of sediment at these points is essential if downstream spawning and rearing areas are to be protected.

To collect data for validation, select a set of non-salmon-bearing tributaries with at least two within each secondary tributary. Other secondary tributaries might be needed to fully represent the range of geological types or landtypes present. Develop or refine erosion curves for landtypes according to local geology, vegetation, and land use categories (grazing, mining, logging). Currently, many of these combinations of inherent characteristics and current condition have not been fully evaluated and some activities such as grazing have still been poorly incorporated into the model.

In addition to validation of the sediment delivery model, validation of long-term application of land management standards, such as institution of sediment delivery standards and riparian reserves, can be evaluated as overall trend in various habitat quality/quantity factors resulting in improvement of salmon carrying capacity. To assess sedimentation in non-salmon-bearing tributaries selected for Level 3 monitoring, map the distribution of LWD for the entire stream length. On an annual basis monitor the volume of sediment storage behind obstructions (LWD, boulders, rock outcrops); pool volumes; V*. Map the location of bars. Survey cross-sections through reaches containing significant lateral and mid-channel bars to estimate annual changes in sediment storage. Map these key areas relative to a fixed benchmark to assess volume changes from detailed surface topography. Sensitive depositional areas also include locations of channel widening and broadened floodplains.

In riffle and glide areas of monitored reaches, assess percentage fines in the bed by depth (McNeil core, dry sieving) as well as surface substrate composition and embeddedness to derive correlations among the methods. Auxiliary methods are free matrix particle (sampled concurrently with embeddedness) during summer/autumn and free winter particle established on a transect through winter habitat. Compare trends in substrate indices in relation to channel type, channel unit type (accounting for gradient and discharge regime), and between non-salmon bearing reaches of the river continuum with salmon-bearing reaches (assessed in other Level 3 work).

In these same tributaries establish stations near the tributary mouths for measuring suspended sediment, turbidity, and bedload. In selected tributaries of the region, install settling basins for use in calibrating standard bedload monitoring data and for more accurately identifying the size composition of material in bedload. Insure that a wide range of tributary watersheds are monitored encompassing a diversity of geological types, watershed topography (relief, drainage density, drainage area, stream profile), and watershed condition (road density, extent of development). Measure peak flows for these streams using crest gages. Calibrate these flows to flows at nearby continuous stage height gaging stations to derive estimates of stream power and flow duration on tributaries. For each major geologic type, establish relationships between various indices of sediment transport and basin development.

In tributary streams assess the accuracy of the stream reach and network temperature models. These models will require accurate input data on percentage riparian cover, canopy gap, water surface width, channel orientation, groundwater input, air temperature, channel gradient, discharge. Although the coarse screening document recommends establishing riparian reserves, monitoring water temperature is useful for validating the temperature model and establishing the linkages between riparian condition and water temperature. In addition, monitoring water temperature provides a sensitive measure of riparian and overall watershed recovery. Experience with applying the temperature model will allow restoration scientists to identify stream reaches where specified quantitative improvements can be predicted if active riparian restoration (e.g., vegetation planting) is carried out.

In selected, representative tributary and mainstem salmon-bearing stream reaches, assess the ability of LWD recruitment models to predict rate of LWD entry to a channel. Monitor levels of in-channel, channel spanning, and floodplain LWD. Record the number and size (length, diameter) of pieces per aggregation or volume. Record the type of habitat associated with the LWD. Monitor regrowth of the riparian vegetation by channel type and geomorphic surface. Riparian data collection should include species composition, percentage cover by vegetative layer, density, height, and DBH diversity of major tree species.

Approach

Apply all data (historical and current) and land system classification mapping (ecoregion, landtype, watershed, riparian, channel type, channel unit type) from a GIS system as the basis for comparing the performances or trends of selected variables central to key models.

Key issues to be addressed

a. In-channel habitat conditions

Apply the same validation monitoring as conducted on salmon-bearing reaches (see p. 55).

b. Watershed habitat conditions

Apply the same validation monitoring as conducted on salmon-bearing reaches (see p. 56)

c. Biotic response

1. Evaluate the use of resident fish populations (sculpins, bull trout, cutthroat trout) as indicators of temperature and sediment trends. Correlate annual population size, community composition, and size class variation with environmental variation.

2. Establish a macroinvertebrate reference collection for streams of different types and condition. Assess the ability to define sediment-sensitive indicators. Relate community structure to habitat stability.

Part IV: Classification Systems in Monitoring

A monitoring framework, to be most effective, should be stratified according to the same sets of principles upon which physical processes and fish distribution are organized. The distribution of land uses across a watershed tends to follow the spatial pattern of land capability. It is a relatively simple matter to identify which land management procedures are applied to which land areas, but more difficult to adequately classify land capability classes. Monitoring of land unit response to a particular management perturbation requires a methodology for classifying land unit types. Likewise, the distribution of fish populations or communities occurs in relation to habitat at various spatial scales. General relationships between the conditions of certain elements of habitat, such as water temperature and substrate fine sediment and fish survival and production, are known with a fair level of certainty for certain fish species (Rhodes et al. 1994), although the complex interactions among certain other habitat factors relative to various temporal and spatial scales make a comprehensive description of optimal habitat conditions for any fish species problematic. For a fish population or community to be sampled meaningfully, a consistent and ecologically based sampling framework must be available that accounts for habitat at various spatial levels.

Similarities in types of land uses occur according to similarities in climate, soils, potential natural vegetation, and topography. In short, ecoregions or subecoregions of similar type (when classified according to inherent natural characteristics) provide conditions suitable for limited sets of land uses. Range allotments in similar ecoregions share similar climate, potential natural vegetation, major soil types, etc. Consequently, monitoring results from representative allotments could be extrapolated to other members of the same class (i.e., those other allotments in the same ecoregion having a similar grazing system and grazing intensity).

The habitat for a fish population can be considered to be the entire system of watersheds and their stream systems that are used during the full lifecycle. During the freshwater development phase in tributaries, the habitat system includes all units that provide for biological functions of adult migration and holding, egg deposition and incubation, fry emergence, juvenile rearing and overwintering, intra-basin migrations, smoltification and emigration. When targeting any of these developmental stages in sampling, more limited portions of this habitat system are identified. Because the watershed contributes water, sediment, nutrients, woody debris and other organic material to the stream system and interacts with the stream throughout its length, the character of fish habitat, at all points along the river continuum, is strongly shaped by the contributing watershed's inherent properties (gradient, elevation, climate, soils, landforms, etc.) as well as the current state of portions of the watershed (e.g., vegetation status of the riparian zone and floodplain; bank condition). A purpose of habitat classification, then, is to make use of what is known about the inherent properties of the stream and its watershed to attempt to identify habitat units with similar capabilities and that may function similarly to changes in ecological processes (whether natural or not). Given this capability, habitat conditions will vary according to current state of the watershed and the stream (i.e., whether the riparian zone has been harvested, the kind of vegetation present and seral stage, condition of streambanks, etc.). When applied to fish populations, this two-stage form of classification identifies a potential and probable current state for fish population performance. Habitat classification and explanation of current states, though, are not simple, because the character of any stream reach is a product of local inherent features (e.g., channel slope, valley morphology, potential riparian vegetation), current state (e.g., existing vegetative state, LWD, W/D), and also the inherent potential and current states of the contributing watershed. Even worse, lags in response of a salmon habitat to the condition of the watershed frequently occur due to either climatic cycles or to exceedance of response thresholds. Cumulative effects from upstream watershed activity can lead to shifts in sediment status of a particular stream reach, channel type, or channel unit type downstream, despite the local conditions (e.g., bank condition). Because the magnitude of this cumulative watershed effect to a downstream channel type can vary with watershed type (e.g., erosive versus non-erosive watershed), the long-term dynamics of fish populations under natural and management-induced perturbations depend partially on watershed characteristics. They also depend upon the downstream mediation of watershed effects (water flow, sediment delivery) by stream characteristics (e.g., gradient, cross-section, channel materials, associated floodplains, and side-channels). For this reason, hierarchical habitat classification spanning the levels microhabitat, channel unit, channel type, to stream system and watershed is an important consideration in establishing sampling strategies that allow comparison of fish populations from habitat units found within similar hierarchical contexts (i.e., having similar environments).

Of course it will probably never be possible to devise a perfect habitat sampling scheme that reflects a population's environment the way it is experienced by the fish. Habitat types are essentially mental constructs of the biologist that express collective experience about the kinds of locations in a stream that are preferred by certain species or life stages. This is reflected in greater numbers of species or various life stages in these areas than in other areas. The habitat construct is also partially a system imposed upon the biota by the researcher. That is, the initial experience in which fish numbers were related to kinds of locations was based upon sampling according to a preconceived spatial scale. That is, the notion that a riffle is a discernible physical unit of channel structure dictates that a riffle be sampled as a whole. This may hide the importance of internal sub-units of habit and further, there is no assurance that one riffle is similar to any other riffle. There might be valid reasons to suspect that the center of a stream would be used differently by a species than the edges, leading to stratification of sampling of the riffle; or a riffle with an adjoining side-channel could be different from one with an adjoining pool. This hypothesis could be validated by sampling, but even so there is every reason to believe that some other stratification would produce an even clearer differentiation among sub-units.

Rather than identify areas of habitat use, one could observe fish use of points on the stream bottom or in a three-dimensional channel volume in terms of percentage of time spent at each point. Then by identifying the physical characteristics of each point one can express habitat utilization. This tends to be a subset of habitat preference because of inter- and intra-species competition and predation effects. The use of one point, though, may depend heavily on the characteristics of nearby points. That is, the preference to hide in a location with low current speed might be heavily associated with presence of nearby points having fast current speed and carrying abundant food supplies or, even more confounding, by the sequence of habitat types and velocities at various scales. Even by reducing habitat to physical characteristics of individual points in space, the full description of what makes each point a location of high or low preference is multifaceted. Identification of a habitat area having uniform characteristics (i.e., made up of points with similar properties, or repeating sets of points with contrasting properties, such as deep, slow water points adjoining shallow, fast water points) could be conceived as a process of agglomerative clustering of points having similar properties or juxtaposed areas of contrasting properties into an area, or taking a large area and plotting contours of preference. In the latter case, sample units might be between contour lines. Regardless how units of habitat are mapped on the stream bottom, the process has as much to do with our concepts of the way the world works or economies of conducting sampling as it does with the myriad of factors affecting fish distribution. Variability of monitoring data on fish communities arises partially from the disconnect between our concept and actuality, from chance in occupation of habitats by individuals of a species, and from natural variation in population size, and from multiple anthropogenic effects which cannot effectively be scaled.

Why use Classification?

Within the context of the CSP, it might not be always necessary to have a classification system in place for purposes of defining sampling units. Monitoring of recovery and trends can certainly be done for entire watersheds or stream systems, fixed stream reaches, fixed channel units, or fixed channel cross-sections. The watershed or stream system is defined by watershed boundaries and should represent a logical salmon production unit for CSP monitoring. Definition of monitoring reaches should be according to sensitivity to management effects or historic use by salmon for spawning or rearing. Monitoring of fixed channel units, such as particular spawning sites, over time is feasible for streams whose channel units do not migrate with channel changes. Cross-sections might be selected according to a stratified, random, or uniform distribution along a selected monitoring reach (e.g., to monitor changes in substrate sediment accumulation) or could be established according to hydraulic principles at strategic points to provide stable sites for monitoring discharge.

Even though status and trends can be mapped or expressed for any arbitrary sampling unit over time without resorting to a classification procedure, questions remain regarding whether the sample units are representative (provided that only a portion of the system is monitored). And if they are representative, what kinds of units are they representative of? If sampling is done according to a complete survey or a uniform sampling pattern, one can express system condition in terms of an area weighted average. If conditions are expressed for separate sub-units of the system, these could be arbitrarily identified (e.g., by subdividing a stream system into 100-m segments starting from a random point) and monitored separately over time to express spatial aspects of recovery. However, it is more useful to stratify the system according to classification principles to establish meaningful physical boundaries prior to initiating biological sampling.

Some proponents of classification for monitoring recommend use of biotic criteria rather than physical criteria to establish boundaries of sample units in the stream. The idea is that the species or community is the ultimate indication of the "natural" physical breakpoints differentiating sample units in the stream. There are several things wrong with this idea. In the Snake River basin many streams do not have the full array of historic fish species present. Those that are present often have their distributions limited by exotic species or overabundance of certain native species. Because current habitat conditions are frequently highly degraded, especially in larger order streams, the species or communities inhabiting any reach presently may indicate little about the long-term capability of the reach. That is, patterns of distribution of fish species and life stages in degraded coldwater systems (influenced by fish whose preference may be for streams of low structural diversity, warm water temperatures, high sediment loads, etc.) do not necessarily suggest "natural" sampling units that would be useful for monitoring long-term recovery trends for salmon. Also, current physical habitat performance (i.e., a snapshot of long-term habitat regime under various natural or management-caused environmental stresses) or dependent biotic performance may not be a good indicator for the long-term capacity of any particular stream system or stream reach to provide certain types of habitat conditions or communities. These capacities may be better judged from more temporally stable features of the watershed or stream (Warren 1979, Frissell et al. 1986, McCullough 1988). For this reason, application of a habitat classification system based upon capacity is apt to provide a more rational system for identifying features that differentiate habitats over time periods necessary for full ecologic recovery.

Benefits in applying classification principles to a monitoring plan may include the ability to provide (1) a framework for comparing fish response in geographic units with common properties, (2) a basis for extrapolating monitoring data from one unit to others with similar properties that were not sampled, thereby allowing estimation of fish population size and species diversity for the entire set of reaches of the class, (3) a means to recognize inherent potential of a habitat system of any scale and the consequent ability to assume various states over the course of degradation or recovery, (4) a means to generate hypotheses concerning fish response to habitat quality or trends in quality by monitoring habitat units of certain types, (5) the basis for selecting streams or reaches having high potential for recovery, contribution to fish production, and for predicting physical behavior, such as bedload-discharge relationships (6) a basis for matching native riparian vegetation used in riparian planting projects to site conditions, (7) a means to assess how many types of sampling units there are in total so that effort can be apportioned relative to the percentage of each habitat type or according to the fish use of each type, (8) a framework for stratifying land types so that the management practices applied to them and the instream consequences can be monitored and effectively compared, (9) a management framework that allows managers to use monitoring results indicating high sensitivity of certain stream reaches or hillslope types to management practices to alter practices on all other similar units, (10) a framework for viewing numerous annual shifts in community composition for a habitat class as a portion of the long-term potential of the class, given a definable environmental regime, or as a way to track the shift in relation to trends in the environmental regime. Even absent a full hierarchical classification of habitat, monitoring of longitudinal trends in condition at fixed points in a stream continuum would allow useful information about effectiveness of recovery efforts and would allow generation of hypotheses concerning reasons for differences in recovery trajectories among watersheds.

Numerous classification systems are available to be applied to monitoring or research studies at various geographic scales or levels of resolution of biologic data. Representatives of the major kinds of classification systems used at various spatial scales in the western US will be briefly reviewed to illustrate the diversity of approaches.

Physical System Classification

Microhabitat

Moore and Gregory (1988a, 1988b) recognized the differential use of stream reaches by young-of-the-year cutthroat trout in high gradient Oregon Cascades streams between stream margins and mid-channel areas. Increased shoreline development of channel margin area resulted in increased cutthroat density and extreme lack of these microhabitats resulted in absence of emergent fry. Bovee (1982) described microhabitat as a product of the rigid structural channel elements and variable hydraulic conditions of flow velocity and depth at a point on the cross-section. Microhabitat for juvenile fish and invertebrates has been described as features within riffle channel units, for example, such as interstitial spaces among cobbles, surfaces of cobbles, surfaces of LWD, velocity refuges downstream of LWD or boulders, shadows on the stream channel bottom from LWD, overhanging banks, roots of riparian vegetation extending into channel margins.

Channel (habitat) units

In this report the fundamental units of habitat will be referred to as channel units, using terminology of Sullivan (1986) and Grant (1986). In fisheries and stream invertebrate literature, habitat has commonly been considered to be the riffles and pools of the channel. These structures are channel features whose boundaries are marked by changes in bedform, channel cross-sectional area, water surface slope, turbulence, and current velocity. Aquatic ecologists often recognize the presence of glides or runs as ecologically distinct units, which are zones of transition between riffles and pools. These channel units are typified by less turbulent flow. Their spatial extent varies with discharge (Hawkins et al. 1993).

Differential use of a stream reach between riffles and pools has long been recognized by aquatic ecologists. Herrington and Dunham (1967) recognized riffles and pools as fundamental habitat units and subdivided pools into five pool quality classes. These classes were defined by pool length relative to average channel width, maximum depth, and degree of cover provided by logs, woody debris, boulders, live vegetation or overhanging banks. In this system a greater amount of cover can compensate somewhat for reduced pool depth within a particular quality class. A similar system of pool quality classification was used by Platts et al. (1983). Bisson et al. (1982) devised a system of classifying channel units according to formative elements, location within the channel, and pattern of water flow through the unit. This system recognizes six pool types, three riffle types (differentiated by gradient, current velocity, turbulence, and substrate size) and glides. Bisson et al. (1982) found significant differences in use of these channel unit types among the three salmonids observed in four western Washington streams (coho salmon, steelhead, and cutthroat trout) and also among age classes of these species. Use of channel units appeared to be related variously to presence/absence of certain other species, presence of associated features such as LWD or overhanging banks, which may or may not be defining properties of the channel units, and the flow velocity, depth, and turbulence characteristics of the channel units. This system made significant contributions to habitat classification methodology by pointing out differences in the rearing environments within different pool types and by applying genetic descriptions of pool formation, which seem to be well related to hydraulic processes. However, the generality of preferences by species or age classes for pool classes needs comprehensive evaluation. It is likely that the hydraulic conditions dominant during formation of various pool classes (e.g., at high flows) provide environments for biota at these times that are significantly different from those existing when sampling occurs. This could result in many pool classes having remnant physical differences allowing their classification after pool formative events that essentially provide little difference in hydraulic condition during low flow. In addition, the tendency to apply ad hoc channel unit characteristics, such as presence of LWD, in explaining degree of utilization implies that fish use of channel units is complex and potentially more effectively stratified by employing other factors.

Hawkins et al. (1993) devised a system of channel unit classification largely based upon the Bisson et al. (1982) system. The Hawkins et al. classification system is a three-level hierarchical system that differentiates fast- and slow-water habitat classes at the first level. Fast-water units are distinguished as turbulent and non-turbulent. Slow-water units are distinguished as scour or dammed, according to their mode of formation. The scouring and dammed natures of pool channel units are related to their tendencies to retain organic matter and fine sediment. Dammed pools are more retentive and are more associated with LWD. Further subdivision of fast-water channel units is made according to the factors of gradient, supercritical flow, bed roughness, mean velocity, and step development. Subdivision of the slow-water channel units proceeds by consideration of their location within the channel, locations of the maximum depth on the longitudinal and the cross-sectional profiles, substrate character, and forming constraint. In all, Hawkins et al. (1993) identified seven types of fast-water channel units and eleven types of slow-water channel units.

Channel type

The stream channel classification system devised by Rosgen (1985) is based on water surface gradient, sinuosity, width/depth ratio, dominant particle size of the bed and bank, channel entrenchment, valley confinement, and landform and soil stability. A combination of quantitative ranges of performance (e.g., sinuosity) and qualitative descriptions (e.g., degree of confinement) for these variables is used to typify the channel types. Major channel types are designated A, B, C, and D. A-type channels all have gradients greater than 4%, sinuosity <1.4, W/D $\leq$10, are well confined and have moderate to steep sideslopes. Variations in these and remaining factors establish ten categories of A-channels. B-channels have gradients of 1.5-4.0%, sinuosity 1.2-1.9. Other features of B-channels are highly variable, resulting in six types of B-channels. C-channels have a gradient of 0.1 to 1.0%, sinuosity that can be within the range of A- and B-channel types but can also be 2.5+, and are in alluvium and typically have terraces. W/D can range from 3 to 30, depending on the particular C-channel type. The other factors are extremely variable among the seven C-channel types, spanning the full range of possible characteristics. D-channels are braided; all other features are extremely different between the two types of D-channels described. For the most part this system reflects four major stream zones along a river continuum: high gradient, low sinuosity streams, with steep sideslopes, well confined and with generally low W/D ratio (A-channels); medium gradient streams with generally moderate sinuosity and medium W/D ratio (B-channels); low gradient streams in alluvium that can have high sinuosity and W/D ratio (C-channels); and braided streams (D-channels).

Sub-types of the channel-types are delineated according to riparian vegetation, organic debris, stream width, flow regimen (ephemeral, intermittent, perennial), bar development, and meander patterns.

Although this system is widely used in the US for numerous purposes (e.g., establishing sediment rating curves; estimating ratios of bedload to suspended load; expressing sediment transport as a function of stream power; establishing a basis for monitoring), it is not clear what the relationships are between the various types of channel units. That is, the continuum from channel type A, B, C, to D is one of generally decreasing gradient, increasing sinuosity, and decreasing sideslope gradient. Other factors vary greatly among channel type variants. There is no obvious hierarchical structure within any major channel type (e.g., C-channels). There is no accompanying biological theory explaining the physical basis for fish response to channel types, why certain channel types would provide similar spawning or rearing environments, or how during channel degradation or recovery, transitions are likely between some of the channel type variants because they are very much alike. It is not clear from the information presented in this document whether these channel types represent inherent, immutable characteristics of a site, with the sub-type criteria expressing simply annual trends in state (e.g., amount of in-channel organic material, riparian vegetation type).

The factors included in Rosgen's channel type classification provide the basis for creating many hypotheses about linkages between fish habitat and fish population response. Juvenile salmon distribution is typically restricted to channel gradients of $\leq$4% (Platts 1974). Channels of high sinuosity provide more diverse hydraulic environments with frequent meander bend pools and velocity refuges. Channel materials are highly related to distribution of salmonids (Nelson et al. 1992). The degree of confinement (or ability of high flows to expand over floodplains and the presence of riparian vegetation on the floodplain to act as flow refuge areas improves habitat diversity and survival during floods (Welcomme and Hagborg 1977, Welcomme 1979). The combination of W/D and confinement define the ability of W/D to vary during high and low flows. A great increase in W/D caused by bank damage and loss of riparian vegetation eliminates deep-water refuges for larger salmonids, reduces depth diversity, and leads to reduced tractive forces on the streambed, local aggradation, and fine sediment deposition. This produces unfavorable conditions for incubating eggs or rearing juvenile salmonids. The ability of stream channels to adjust W/D in relation to streamflow regime depends in part upon channel and bank materials. Textural composition of alluvial materials or streamside soils determines the ability to form overhanging banks. The influence of the multiple physical factors that define any individual Rosgen channel type also provides a template for characteristic habitat development. These multiple factors used to define the Rosgen channel class may have varying levels of influence on habitat development and biotic distribution, making it difficult to attribute biotic distribution to particular channel-forming factors.

Stream size (width or drainage area) is a significant factor influencing biotic distribution that is used in the Rosgen channel type classification only as a sub-type criterion. Channel gradient may on average be related to drainage area but reaches of similar gradient can have very different drainage areas. This point illustrates a reason why monitoring according to the basic delineation of channel types may omit factors of great biological significance that will produce great variation in monitoring results.

Rosgen's (1985) widely used channel type classification system was refined to include 7 major types that differ in entrenchment, channel gradient, W/D, and sinuosity (see Rosgen 1994). The revised classification eliminated the overlap in delineative variables. This makes classification more definitive, but brings into question the strength of the tendency of any particular stream type to behave within the range of the variables used to identify it. Rosgen (1994) provides examples of progressive channel evolution from one type to another. The degree of stability associated with any particular form is not clear; also, it remains to be learned which transitions are common for a stream type of a certain potential and how various changes in controlling variables can lead to different types of transitions. It is also unclear whether the potential for a certain set of stream types can be classified by inherent geologic or geomorphic features of the landscape.

The revised Rosgen (1994) system identifies A, B, C, D, E, F, and G stream types. These 7 types are each subdivided into 6 classes according to median particle size diameter of channel materials. The median particle size diameter (D_{50}) is called the dominant particle size. This should be distinguished from its usage in many monitoring programs as the most common large particle class. The D_{50} is determined from a cumulative particle size distribution that incorporates fine sediment fractions. Rosgen's 6 particle sizes (1=boulder, 2=cobble, 4=gravel, 5=sand, and 6=silt/clay) produce a total of 42 major stream types. Rosgen showed the utility of his classification system in effectively distinguishing stream characteristics such as channel roughness, hydraulic geometry relationships, and sediment rating curves.

Valley type classification

As part of his channel type classification, Rosgen described the landforms and soils adjoining the channel. The landforms and soils provide information on the water and sediment delivery system of the reach or contributing watershed. Steep sideslopes generally lead to a high degree of entrenchment and confinement of the channel, although fine textured alluvium can lead to deep entrenchment and terrace development. The interrelation between channel conditions and landtypes is something recognized earlier by Platts (1974). He used a geomorphic basis for classification to explain the distribution of salmonid species and density.

Cupp (1989) enhanced the description of valley type classification. He took valley landform to be a primary determinant of channel constraint, which controls distribution of channel units. Typically, constrained stream reaches have high gradient channels and step-pool configuration, whereas unconstrained reaches are lower gradient and have channel units bordered by floodplains and terraces. Cupp identified 19 valley segment types in forested watersheds of Washington. These types were described by valley bottom slope, sideslope gradient, ratio of valley bottom width to active bankfull channel width, channel pattern (constraint; longitudinal form, i.e., straight, meandering, braided; and vertical form, i.e., stairstepped), adjacent geomorphic surfaces, and location within the watershed. This classification system appears to be less ambiguous in its implementation than Rosgen's system because it is more explicit in its description of sideslope gradient and valley bottom-to-active channel ratio. The latter term appears to be easier to assess than

Rosgen's more qualitative descriptions of entrenchment and confinement. In addition, it does not attempt to deal quantitatively with so many terms, such as W/D and sinuosity, thereby making it seemingly easier to fit a reach within a valley type class. Also, it does not attempt to characterize substrate type within the reach. Substrate presumably is subsumed by valley bottom slope, sideslope gradient, and adjacent geomorphic surface.

Land system

a. Land Systems Inventory

The Land Systems Inventory, developed by Wertz and Arnold (1972), has been applied extensively to national forests of Idaho. The theoretical basis for this classification system is that geology, soils, hydrologic character, and climate determine climax plant communities and landforms. Mapping of landforms from aerial photos, likewise, identifies portions of the watershed with predictable soil properties and vegetation. Given a particular landtype, having common soil properties, climate, and landforms, one can predict soil erosion hazard, timber productivity, and hydrologic response. The Land Systems Inventory is a hierarchical classification of the landscape having seven spatial levels, with landtype and landtype phase being the lowest levels. Landtypes are units of 0.1-2.0 mi^2 that have a visually recognizable landform (relief, slope shape, geomorphic processes responsible for shaping the land surface) and certain soil and climax vegetation community properties. Soils are classified at the family level of taxonomy and recognize geologic origin in their description. Plant communities are classified according to the Daubenmire "habitat" system. This is a means of identifying climax vegetation potential with respect to landform and climate.

Lithology and climate are considered accessory components of the landtype. Climate is characterized by mean annual precipitation, percentage of precipitation as snow, 15-min maximum rainfall intensity, mean annual air temperature, soil moisture stress seasonality, and potential evapotranspiration. These regional expressions of climate are extrapolated to individual landtypes based upon slope, elevation, and aspect.

Once a landtype is identified, a land manager can use this classification to aid in predicting the response of the land surface to logging or roading. Sediment delivery efficiency varies by landtype according to the characteristic properties of the landtype, including slope shape, percentage of the landtype drained by low order streams, mean slope of the landtype, soil depth, and soil internal drainage.

The Land Systems Inventory classification incorporates land units that may include entire watersheds but watershed boundaries are not criteria used to delineate a collection of landtypes.

b. Regionalization

Bailey (1976, 1980) developed a land classification system that is essentially a top-down hierarchical regionalization. Like the Land Systems Inventory it is not based upon watershed boundaries. At each level in the regional hierarchy a different major variable is used to identify homogenous landscape areas. Landform, potential natural vegetation, climate and soil are among the variables considered in this progressive subdivision of a region. This system has been criticized by Hughes and Omernik (1982) because it fails to adequately relate regions at any scale with stream water quality.

The "ecoregion" has been reported to be a useful tool for separating geographic regions having considerable differences in stream and lake water quality (Omernik 1987, Gallant et al. 1989) and fish distribution (Hughes, Rexstad, and Bond 1987, Hughes and Gammon 1987, Lyons 1989, Rohm et al. 1987, Larsen et al. 1986, Whittier et al. 1988). Proponents of the ecoregion approach support the hypothesis that characteristics of any ecoregion are clearly expressed in streams contained within the ecoregion and that the similarities among streams within the region are greater than between regions.

Ecoregions, according to the procedures of Omernik and Gallant (1986), are identified in a top-down, hierarchical classification by applying the criteria, potential natural vegetation, landform, soils, and land use at each level. Portions of the landscape within any relatively homogenous region that have all four of the primary delineative criteria expressed are considered to be "most typical" areas; those with only three criteria expressed are "generally typical. Other areas are atypical. By this means distinct regions are identified as having, for example, major differences in combinations of the primary criteria; also, typical and generally typical areas are identified within each delineated region. Presumably streams of similar drainage area would be compared within ecoregions.

Even though the ecoregion methodology allows progressive subregionalization to lower hierarchical levels, practical applications of this subdivision have been limited with respect to aquatic resources. Gallant et al. (1989) state that ecoregionalization, in their view, should proceed qualitatively, recognizing that driving characteristics vary in importance across a region. This might require that soils be emphasized in certain locales and landform elsewhere. Also, they state that regionalizations may need to be customized for each particular application. That is, a regionalization useful for soil erosion might be different from one designed to reflect water chemistry. Ecoregions are not based on watershed boundaries because proponents believe that fish community distributions are more attuned to physical criteria defining ecoregions than to watershed boundaries. This system is reported to be a better integration of land and water systems, but the exclusion of watershed boundaries and their consequent hydrologic implications make this claim dubious. It is known that fish species and subspecies distributions are frequently sharply distinguished by topographic ridges separating drainages, reflecting separate biogeographic history (Li, personal comm. 1994).

It is questionable how fish communities can be managed efficiently if the ecoregions, reflecting factors responsible for fish distribution, are different from ecoregions used to describe homogenous sedimentation environments, hydrologic environments, etc. Two different ecoregions might have similar sedimentation environments (e.g., sediment delivery, transport, particle size distributions with position on the river continuum), but because of numerous other physical differences, the environments for fish are different. Although these two ecoregions could be merged in a sediment-based classification, it would seem more logical to recognize the full suite of physical and biological factors at work and create an ecologically-based classification rather than attempting to select single factors presumed to be most important locally, thereby creating different regionalizations for each purpose. Such a classification principle seems to violate the stated intent of

Omernik and Gallant (1986) to apply a more integrated classification (multiple variables) in creating their first level of ecoregionalization.

Integrated Hierarchical Ecosystem Classification

Frissell et al. (1986) presented a hierarchical system for classification of streams that includes the levels of stream, segment, reach, pool/riffle complex, and microhabitat. The stream system is comprised by the entire surface water system within a watershed whose boundary is defined by surface topography. Using a systems theory developed by Warren (1979) they proposed that a stream system is classified by its watershed class (the environment of the stream system) and certain key factors relative to the stream system itself (slope and shape of the longitudinal profile, and network structure). The watershed is classified according to its biogeoclimatic region, geology, topography, soils, climate, biota, and culture. The indices chosen to characterize a system at any scale are selected to represent system potential capacity. Realized expressions within the context of this potential occur in response to performances of the higher level environmental system and also to impacts within the system itself.

The stream habitat classification system proposed by Frissell et al. (1986) explicitly embeds the stream system with its watershed. Each hierarchical level within the stream system classification can be described in terms of physical boundaries and relative time frame for persistence. In addition, each stream system level has a potential capacity that is a product of major geomorphic or climatic events. Potential capacity then undergoes progressive change attributable to erosional and vegetative processes that results in realized capacities. This concept conforms to the paradigm of dynamic equilibrium punctuated by sudden shifts in metastable states (e.g., see Schumm and Lichty 1965). In this view, stream communities are definable relative to a hierarchical habitat template in which the unit of "habitat" is any level in the stream hierarchy considered.

Frissell et al. (1986) provide examples of stream segment and reach classifications. Segment classes are distinguished by stream order, slope gradient, sideslope gradient, and geology in their hypothetical classification. Segments can be qualitatively described in geomorphic terms, such as steep headwall tributaries or lower valley mainstems. This kind of geomorphic description is also reflected in ordination plots for segments having various positions in the landscape using variables such as segment slope and Shreve link number. Ordination via segment slope and link number tends to confirm that qualitative descriptors of segment type identify relatively discrete groupings of segments according to more quantitative expressions. However, it is largely by definition that steep headwall stream segments will have relatively greater slope and lower link number than that for large stream segments. Frissell et al. also employ ranges of channel gradient to identify the classes. The classes appear to be formed fairly arbitrarily. That is, there is no theoretical physical basis expressed for the classes used, such as thresholds for various physical processes or biotic distribution. For variables such as geology, classes are expressed categorically in terms of major lithologic type. Such a classification tool can be used to compare, for example, sandstone lithology to siltstone, basalt, granite or other rock types. That is, the differences between sandstone and siltstone may be slight with respect to their abilities to form soil, contribute to hillslope instability, produce surface erosion, etc. when contrasted with basalt, so with respect to these physical capabilities, lithology can be a synthesizing principle. Interpreting physical capability from combinations of lithology, climate, and vegetation requires detailed understanding of watersheds and could help to aggregate classes based more on capability than merely using simple categorical information (e.g., lithology, vegetation type, etc.). Further, there is no method described for classifying complex lithology (multiple rock types and complex distribution patterns) with respect to a segment. This same observation holds for the remaining two segment classification variables recommended, potential climax vegetation and soil associations. No method is given for employing these variables when the distribution of vegetation or soils is spatially heterogeneous in the terrain contributing directly to a segment.

The Frissell et al. system does provide a list of key classification variables that can then be interpreted by the user to distinguish classes at any level of resolution. Monitoring biotic response of segment classes, however described, then provides data, collected according to a theory of system capacity, that can be used to verify utility of the classes or to modify the classification system.

The Rosgen system for segment classification employs ranges of slope and other factors to categorize stream channels, similar to the system of Frissell et al. (1986). Rosgen assigns labels to the classes described via multiple variables. His system differentiates channel types within major channel classes that are essentially a function of position in the drainage system. Rosgen's system appears, on the surface, to have general applicability; that is, the classes described according to ranges in major classifying variables can incorporate variation in climax vegetation. Rosgen's basic classes, then, can be found across the spectrum of vegetation types. A limitation to the Rosgen system then is that the physical or biologic behavior of stream channels can only be compared if the "vegetation type" category can be translated into more basic properties (e.g., ability to produce similar equilibrium LWD volumes in channels).

Reaches are classified according to the environment of the reach (i.e., the segment class, that in turn is classified by its environment, the stream class, etc.) and the characteristics of the reach itself (bedrock slope, morphogenetic process, channel pattern, local sideslopes, bank composition, and riparian vegetation condition) according to Frissell et al. (1986). However, in a further illustration of reach classification, these authors used certain considerations such as erosional/deposition properties, dominant geomorphic elements defining the reach, morphogenetic processes, developmental trends, and persistence. Although these concepts are highly meaningful if properly considered in the field, their application requires great skill and understanding of stream mechanics. The seemingly large number of combinations of all factors involved in classification makes a potentially large number of classes. This classification system serves as a vocabulary for describing reaches, but the need for clearer procedures for examining reaches (e.g., identifying whether a reach is erosional or depositional) makes the current system largely a blueprint for further development. Erosional/depositional distinctions, for example, must imply more than whether a reach has a riffle versus a pool, has bars and obvious fine sediment, or has a gradient less than a certain percentage; otherwise, a simple gradient classification would suffice.

The pool/riffle system (Frissell et al. 1986) is a subsystem of a reach comprised by various classes of pools and riffles. The proportion of these pool and riffle units can change within the reach over time. The reach, whose characteristics respond less rapidly than those of its individual component riffles or pools, maintains its class while internally it changes in relative frequency of riffle and pool classes. Riffle and pool classification relies on the procedure of Bisson et al. (1982). Microhabitat types are described in terms of substrate type, water depth, and velocity. The environment for the microhabitat is a riffle or pool class.

The classification system of Frissell et al. (1986) is least specific in description of how to classify watersheds and stream systems, the environments for all the lower level systems from microhabitat to reach. For example, watersheds are classified by their environments (biogeoclimatic region), geology, topography, soils, climate, biota, and culture. The exact expressions of these variables are not given (e.g., what aspects of topography should be emphasized; what are the best indices of these variables to represent capacity)?

McCullough (1988) developed stream classification concepts and methodology in a watershed perspective, also based upon the extensive work done by Warren (1979). This hierarchical system of classification steps down from ecoregion, watershed, subwatershed, stream system, segment to reach. In later work McCullough (1990) described a progression of landscape units corresponding to the progressively smaller stream units down to reach level. Watersheds or subwatersheds subsume their entire stream systems. They also have a series of valleys and hillslopes that are specific to stream segments. A valley contains riverine/riparian ecosystems that are specific to reaches.

In this hierarchical classification (McCullough 1988), watersheds are classified according to geology, topography, soils, climate, and water relations. Each of these major categories of variables are composed of specific variables (qualitative and quantitative). Topographic variables include the slope, aspect, and altitude for all slope facets (standard unit of map surface area) comprising the watershed. In addition, watershed topographic capability is identified by hypsometric integral, drainage density, basin orientation and slope, and spatial distribution of rock and soil type. Watershed climate is classified in terms of the regional climate (watershed environment) and the spatial distribution of solar radiation intercepted by each slope facet on the watershed, accounting for topographic shading. Water relations include hydrologic variables for surface and groundwater that set statistical limits to behavior of the stream system. Vegetation capacity is defined according to the polyclimaxes associated with each slope facet, where the facets have slope, aspect, altitude, and solar radiation characteristics that also are linked to characteristic soil types, given the rock type or residual lithologic material underlying the slope unit. The cultural capacity is constituted by the management history; variables include features such as the miles of road, spatial location of roads, harvest distribution, etc.

The quantitative and qualitative variables associated with each slope facet for every watershed of a given drainage area can be analyzed by discriminant analysis to produce groupings of watersheds in multivariate space representing similarities in characteristics. Alternatively, by reducing the quantitative data for all slope facets of each watershed to statistics (e.g., mean, coefficient of variation, skewness, kurtosis or factors such as slope gradient, elevation, solar radiation; drainage density; hypsometric integral, etc.) for the watershed as a whole, these data can be used to produce multivariate classification of the watersheds (e.g., via clustering). The two approaches tend to produce similar groupings of watersheds that can be inferred to represent distinctive classes. As in the stream classification of Frissell et al. (1986), watershed classification stratifies watersheds according to variables that are taken as surrogates for potential capacity. Within a class so-defined, management and natural events create a realized capacity (a subset of potential) that defines a range of probable system behaviors in relation to environmental condition. In less theoretical terms, for example, a watershed that is inherently prone to landslides because of geology, climate (high rainfall events), and topography (steep gradient sideslopes) may have a tendency for a low number of widely scattered landslides during 100-year rainfall events in pristine watersheds, but after heavy development, landslides may be numerous, more frequent, and more severe for many decades. In-channel sediment regimes and channel morphology shift so that the stream system, now under high sediment input rates, develops a different bedload transport relationship with water discharge.

Biotic Classification

Riparian plant communities

Kovalchik (1987) developed a riparian vegetation classification to allow prediction of plant community successional trends on disturbed riparian sites. This classification system of late seral and climax riparian associations and community types is based on physiography and riparian landforms. Physiographic variables considered are climate, geology, and geomorphic processes. Riparian landform is described by the variables elevation, valley gradient, fluvial processes, water regime, valley cross-sectional morphology, valley width, and soils. Climate is linked to vegetation species composition, annual precipitation, and hydrologic regime. Geology determines drainage pattern and soil types found on riparian surfaces. Soil is classified to suborder using SCS (1975)(as cited in Kovalchik 1987). Floristic features that can be described for riparian associations and community types include cover by tree, shrub, graminoid, and forb as well as percentage cover by dominant species and growing site condition.

In further attempts to classify riparian communities of central Oregon, Kovalchik and Chitwood (1988) used four hierarchical levels: physiographic area, watershed, riparian landform, and fluvial surface. Their physiographic areas were based on geology, climate, and stage in evolution of the landform. Watershed differences in order, aspect, elevation, geology, soils, sediment transport, water regime, and vegetation were considered essential in riparian management planning. Riparian landforms were characterized by valley gradient, valley width, elevation, fluvial processes, and soil parent material. Fluvial surfaces were classified as active floodplains, terraces, channel shelves, streambanks, overflow channels, soil texture and structure, and water table.

Stream communities

Particular stream segments have been identified by the predominant fish species occupying the zone (Kuehne 1962, Hawkes 1975, 1977). Such zonation typically follows the progression from high gradient headwater streams, moderate gradient montane streams, to low gradient, meandering streams on low elevation floodplains. Classification of stream segments by biologists traditionally has not emphasized hierarchical classification but has focused on relatively simple indices such as channel width, slope, mean depth, and order (Ricker 1934, Huet 1959, Whiteside and McNatt 1972, Vannote et al. 1980). Although stream communities can shift composition gradually in a downstream direction, as expressed in the river continuum concept (Vannote et al. 1980), major shifts in community composition also occur as large tributaries enter a mainstem river or where there are major transitions in landform, lithologic type, gradient, or a combination of these (Bruns et al. 1984). The influence of the mainstem on community composition of the lower reaches of a tributary has also been observed (Fausch et al. 1984). Hughes et al. (1990) and Fausch et al. (1984) caution that substantial differences in fish assemblages can occur between sites on streams draining one versus two ecoregions or topographical regions. Downstream trends in community composition must not be assessed by compositing samples from different channel units (e.g., riffles and pools) because pools, for example, may be degraded more severely than riffles. Consequently, downstream sampling only of riffles may not be sensitive to trends in degradation or recovery and composited samples may mask the localized impact (Kerans and Karr 1994).

Some authors (e.g., Cushing et al. 1983) prefer to view aquatic macroinvertebrate communities as existing on a habitat gradient rather than in discrete, identifiable habitat units. Such a view does present a useful perspective in consideration of continua spanning several stream orders in a stream system. However, proponents of this view typically sample multiple cobbles (microhabitats) within a riffle to describe the macroinvertebrate community of the riffle (Cummins 1994). The cobbles are considered to represent replicates for the riffle. Consequently, even in the description of community composition at a point on the continuum, a riffle is considered to be a relatively homogeneous site having multiple sampling points that express its biotic characteristics. So, embedded within the continuum view are hierarchical stratification concepts (reach, channel unit, and microhabitat) based upon assumptions about relative homogeneity along the continuum.

Rationale for Emphasis in Monitoring of Selected Portions of the Landscape

Location of monitoring sites is critical in various ways to developing an understanding of watershed and stream condition. Monitoring, as recommended in this document, is focused at several spatial levels with varying intensity and analysis in response to different issues. Efforts to detect early warning of degradation are focused in tributaries near projects. At a more intensive level of monitoring, reaches in tributaries are also the focus of effectiveness and validation monitoring. Monitoring also is conducted in non-salmon-bearing tributary reaches to document cumulative effects of land management activities. These tributaries are relatively reactive geomorphically and in-channel effects can generally be traced to sources. Data from salmon-bearing reaches are aggregated at the watershed level to elucidate long-term trends in in-channel habitat characteristics throughout the watershed that arise from cumulative effects but which are more difficult to trace to a discrete activity or suite of activities within salmon-bearing reaches. Habitat data are also collected at greater intensity concurrent with salmonid monitoring to elucidate linkages among habitat conditions and salmon response, as well as providing higher resolution trend data for both effectiveness and validation (calibration exercises that allow analysis of trends in a spatially disaggregated fashion. Such data can be used to investigate reasons for similarities/differences related to land use/reach/or watershed characteristics (or combinations thereof).

Monitoring sites can be located spatially according to recommended hierarchical classification procedures emphasizing a combination of geomorphic and habitat-type strata. These strata can differentiate headwater tributaries, non-salmon-bearing tributaries, and salmon-bearing tributaries; lowland, meandering rivers and upland, high gradient streams; pools and riffles; thereby, representing position in the watershed and also channel morphological features. Single factor monitoring of, for example, water discharge or water chemistry may need no more classification of the monitoring site than to identify drainage area, local channel gradient, and channel cross-sectional morphology. A hierarchical classification system, though, provides a unifying framework for monitoring of all factors and examining the interrelationships of factors within and among streams.

Monitoring sites are also selected variously for their stability or sensitivity to change. Examples of site selection according to relative stability or sensitivity follow.

Microhabitats

In headwater streams to salmon-bearing streams, microhabitat quality trends are sensitive reflections of upstream activity. For example, sediment deposition in cobbles (embeddedness), deposition as wedges behind boulders and large woody debris, and deposition in bars, channel margins, side channels, and backwater areas provide indicators of watershed effects that alter sediment transport dynamics and salmonid production and survival. Loss or creation of channel bank overhang represents significant change in available rearing areas. This is associated with change in bank stability and is linked to riparian and upstream condition. Monitoring of this microhabitat at a channel unit provides information on suitability for summer and winter rearing.

Headwater tributaries

These small order tributaries flowing out of project areas deliver sediment and heated water directly to larger streams. Their small size makes them especially amenable to measuring suspended or bedload sediment delivery. These sites may provide the clearest signals for localized impacts from forestry activities due to (1) the availability of comparable reaches in unperturbed areas to serve as controls (2) proximity to activities, limited dilution of the input with distance from the source, and limited lagging of the in-channel response after the input or initiating event; (3) reduced overprinting from different effects in different parts of the watershed; (4) greater homogeneity in watershed environmental characteristics.

Stable cross-sections

Sampling at fixed sites is often done for purposes of taking water temperature, fine particulate organic matter transport, water chemistry, and suspended sediment samples in conjunction with continuous water discharge measurements. These sites are selected for relatively high channel morphological stability so that reliable rating curves can be established. Cross-sections for alluvial reaches are less stable than those for bedrock controlled reaches and need to be remeasured more frequently or else reinforced by rigid structures. Stable cross-sections may be associated with certain channel units that maintain their position in the stream system under various geomorphic events. For example, monitoring trends in residual pool volume can be conducted on a per mile basis within a reach type, on a watershed basis, and also on individual pools that occupy strategic positions in the stream system. Such pools may have biologic importance as salmon holding pools and may be geomorphically stable in their physical location because of channel constraining bedrock (channel narrowing or bending) or lithologic transitions producing waterfalls, both of which cause pool formation.

Sensitive cross-sections

Parts of stream reaches that are sensitive to change with increases in discharge are monitored to evaluate degree of perturbation to the stream system with management induced change in input parameters (discharge, sediment). Monitoring techniques include changes in cross-sectional profile to show lateral scouring; use of scour monitoring chains; bedload transport estimation. In meandering, lowland, salmon-bearing stream reaches, cross-sections of greatest reactivity are at meander bends. Intensity of sediment deposition in associated point bars is linked to intensity of local scour and channel migration. Overall, for salmon-bearing stream reaches of the Columbia River basin, the influence of depositional processes is strong, affecting the bed substrate composition of riffles and pools on a consistent year-long basis and on seasonal cycles. The sensitivity of channel cross-sections can be greatly increased by removal of bank stabilizing vegetation. In alluvial streams, pool locations are relatively mobile compared with stream reaches having greater bedrock control. Therefore, reach monitoring of changes in areal proportions and volumes of channel units within the reach may be more revealing of habitat alterations affecting the fish community than morphologic changes at selected cross-sections.

Sensitive reaches

Certain portions of a river continuum are more sensitive to change (less resistant) or are less able to recover after change has occurred (less resilient) than others. This makes certain areas more useful as early warning monitoring sites and other areas useful as indices to the time period for full system recovery (i.e., recovery of full function of the least resilient river system component). Sensitivity depends upon the factor monitored. That is, a single reach type is not necessarily sensitive to all perturbations. The sensitivity of stream reaches varies by channel type according to management effect. For example, A-channels in unconsolidated materials of cobble and smaller particle sizes are very prone to increasing channel erosion due to vegetation loss and increased peak flow and have a low recovery potential (Rosgen 1994). C-channels subjected to increased vegetation loss, sediment delivery, and peak flow respond by widening and deposition. C-channels with dominant particle sizes of gravel or smaller are highly sensitive to these perturbations but tend to have good recovery potential (Rosgen 1994). Small sluggish streams have the greatest unit increase in temperature per unit length and percentage vegetation removal. Mainstem reaches are relatively resistant to temperature increases from a single small stream or from riparian vegetation removal along their banks but are sensitive to cumulative effects of temperature increase in numerous tributaries. The stability and orientation relative to channel direction of LWD in small streams may not be as greatly affected by a unit decrease in the size of LWD input as in a large stream.

Sensitive habitat components, such as reaches, that support certain species may have the ability to dramatically affect the survival, persistence, and spatial distribution of the species. If the species populations are sensitive and they depend directly upon habitats that are sensitive, their survival may be threatened. Conversely, a reach that was degraded and is not highly resilient may present a critical survival problem to a sensitive species. Sensitivity of a species can be attributed to factors such as (1) extremely small population size making the population subject to extreme decreases caused by stochastic environmental or management effects, (2) the species occupying habitats that are at the edges of its historic distribution, and (3) the species depending on habitats providing conditions that place the species at a bioenergetic or behavioral disadvantage relative to other species. Management-induced habitat changes that increase the susceptibility of a species to population fluctuation can involve restriction of habitat size, habitat connectivity, uniform reduction in quality, loss in quality within selected components of the habitat system (which may cause a production bottleneck or elimination of a particular life history attribute), or major alterations in material or energy input regimes (variations in magnitude, intensity, and duration of inputs).

Certain reaches located within generally degraded salmon-bearing tributaries provide the best remaining habitat areas because they confer environmental stability. A good example of this situation exists with a stream system that has experienced widespread riparian vegetation loss. Where these streams meander through meadows, riparian vegetation is frequently also sparse owing to grazing. However, the elevated water temperatures flowing through the reach can be moderated at point locations by groundwater inflow near channel banks or on channel bottoms. Where this inflow can be temporarily trapped in channel margins or in deep pools, thermal refuges are established that moderate diurnal temperature variation. Because remaining thermal refuges are often limited in area, they could be considered a stabilizing habitat factor, but one that is sensitive to further habitat degradation because the means for loss of these qualities can be from basin-wide activities as well as local impact [e.g., pool loss caused by sedimentation (see Frissell 1992) or loss of LWD or riparian vegetation; bank damage; increased drainage of a meadow in agricultural development or road building].

Sensitive riparian areas: Variable source areas

Riparian areas in general are a sensitive terrestrial portion of a watershed comprising a significant element in fish habitat. Alterations in structure of riparian vegetation, soils, or landforms have effects on fish habitat conditions that are disproportionately significant considering the relatively limited area they occupy in a watershed. Riparian vegetation and soil conditions in floodplains provide a sensitive linkage between terrestrial and aquatic systems. Floodplains alternately store and deliver water to stream channels through rising and falling portions of the hydrograph on an individual storm as well as annual basis. This process adds stability to the flow regime. Riparian and floodplain activities that alter this process can likewise alter the discharge regime, channel characteristics, and area of habitat available at critical times. Floodplains or wetland streamside areas are significant landforms associated with variable hydrologic source areas (Lee and Delleur 1976, Parizek 1978). The size of the source area varies dynamically in relation to antecedent moisture and rainfall characteristics and is also a function of topography and soils. Mapping of these hydrologically active areas may be possible by analysis of topographic concavity, producing terrain with convergent subsurface flows (Heerdegen and Beran 1982). These source areas transfer precipitation, nutrients, and stored sediments rapidly to channels. Freeze-thaw cycles in riparian soils are enhanced when riparian vegetation is removed, contributing to increased erosion and sediment delivery (Bohn 1989). As soils become saturated in floodplains, frequency of overland flow increases in these source areas. This increases the likelihood of sediment delivery to the channel, an effect that is exacerbated with vegetation removal.

Sensitive hillslopes

Hillslope form and position on the slope are frequently associated with erosional processes and hillslope evolution. Models of sediment delivery are frequently heavily dependent upon factors such as slope gradient, soil depth, soil type, bedrock dip, and precipitation. Risk of surface or mass erosion are based upon this combination of inherent characteristics and also land use impacts (presence of roads, road construction method and drainage, vegetation removal). Toeslopes impinging upon stream channels may provide sources of coarse or fine sediments during channel scouring events, especially when these slopes are part of active earthflows or exist on outside channel bends. Stream headwall areas are frequently a source of stored sediments and organic material that enter channels in debris flows. Release periodicity and magnitude for these sediments is altered when these material source areas are logged and roaded. Frequency of debris flows decreases with decrease in slope steepness. The upper extent of the stream network, consisting of low order perennial and intermittent channels can be a sensitive portion of the landscape to monitor.

Sensitive stream networks

Stream networks vary in their drainage density, a feature that can cause differences in hydrologic response, including ability to receive and transport water and sediment. Network structure provides a description of the branching pattern. Networks that are trellis versus dendritic in structure have been shown to have different rates of rise to peak flows. This structure is generally geologically controlled. Drainage density can be considered as a parameter defining capacity for hydrologic response, or may also be a product of the interaction of water with the landscape, vegetation, soils, etc. Changes in drainage density can occur in sensitive landscapes subject to headward erosion as upper portions of the watershed are harvested.

Sensitive watersheds

Watersheds vary in certain inherent characteristics that make them respond differently in hydrologic processes. The hypsometric integral expresses the distribution of basin area with elevation. Watersheds having a high proportion of their area at high relative elevations may exhibit a greater intensity of erosional processes characteristic of the relatively higher precipitation, reduced vegetation, and increased slope steepness. Statistical measures of average basin slope as well as drainage density indicate a capacity for erosional processes and sediment transport. Watersheds having extensive, widely distributed floodplains with permeable alluvium have less interannual variability in low summer flows than watersheds without this buffering capacity. Even greater interannual low flow stability is found in glacial-fed streams without floodplains or in spring-fed streams. Watershed orientation is likely a significant differentiating characteristic with hydrologic importance, especially in snow dominated regions. It alters the proportion of slopes with north versus south aspects which exerts a powerful control on the energy budget which strongly affects snowmelt magnitude and timing and evapotranspiration.

PartV: Critical Analysis of Selected Parameters

Watershed Stability and Erosional Characteristics

Qualitative analysis of watershed stability by reconnaissance monitoring

In the Snake River Subbasin, two of the most important, interrelated variables that affect salmon habitat are the maintenance of watershed stability and magnitude of sediment delivery. They are especially germane for land units within the Idaho batholith. For several decades, Forests within the batholith have had to deal with erosional consequences of developing this sensitive geological formation. The case history of the South Fork Salmon River is an excellent example of watershed instability, severe sediment delivery problems and attendant reductions in salmon populations and survival (Platts et al. 1989). Idaho Forests (including some outside the batholith) have developed methodologies, models, and techniques to assess watershed impacts on stability and sediment delivery. Some of these techniques have been in use for over 15 years.

Watershed stability is grounded in the concept of ***"dynamic equilibrium"*** (Patten 1989, Heede 1975 and 1980, Morisawa 1981) in that within a certain range of intensity and frequency of disturbance events, the streams respond by shifts in structure or condition and tend to return to initial conditions after disturbances. Permanent changes in watershed condition may lead to permanent shifts in chronic disturbance or periodic effects of such magnitude that the stream system may shift into a new equilibrium. This may involve a change in physical or biotic capability (i.e., a change in ecological integrity).

Evidence of the magnitude of disturbance, which can lead to growing watershed instability, can be assessed in terms of frequency and extent of major events such as: mass erosion, debris/mud flows, major flooding leading to channel braiding and widening, major alteration of flood frequencies, accelerated scour and fill, major shifts in bed composition, extensive channel bank erosion, and other channel changes (Patten 1989, Heede 1975 and 1980). A reconnaissance-level inventory is needed to identify and document such events.

Sediment delivery

In Idaho, several techniques are used to estimate sediment delivery. Idaho Forests have been using the USDA Regions 1 and 4 Sediment Model (USFS 1981) to estimate sediment yields from forested watersheds—both natural and management-induced components. Individual Forests have attempted to calibrate it to local conditions. The Boise NF has produced the BOISED model, the Nez Perce NF the NEZSED model and the Clearwater NF the WATBAL model. These Forests have used the models to assess the potential of their timber and logging road projects to produce sediment and alter stream channels. The Clearwater NF uses the idea of ***geomorphic threshold*** to assess the stability of streams (Patten 1989). The geomorphic threshold is an estimated level of sediment delivery at which major changes in channel morphology are expected. At far lower sediment delivery levels, substrate changes occur. WATBAL estimates the geomorphic threshold for each watershed and expresses it in terms of percent increase of sediment yield over natural that the system can be expected to withstand for a short time without any major readjustment to the stream channel. The threshold was based on landslide data collected on the Clearwater NF and some research watersheds in the southern Idaho batholith (Megahan et al. 1978). This information was also used to update and validate the mass erosion component of the WATBAL model. While experience with the model's predictions has substantiated its utility and effectiveness at assessing sediment delivery, the assumed thresholds of sediment delivery at which substrate and channel changes occur are entirely suspect. Severe degradation and preclusion of recovery in degraded habitat has occurred at sediment delivery levels far below those assumed to cause damage, as case histories indicated (Rhodes et al. 1994).

Sediment and water yield models have continued to evolve for use in forested environments of the Pacific Northwest. Region 1 of the Forest Service (Idaho/Montana) has recently produced a second generation version of WATBAL (the Clearwater NF's model; USFS 1993) called WATSED. Region 1 has directed all its Forests in Idaho and Montana to adopt and implement WATSED. It is very likely that other Forests and BLM units within the Snake River Subbasin will use WATSED.

WATSED is a data-intensive tool. A land system inventory is the primary and requisite data source for the model. This inventory contains basic landtype data for a designated area. Each Forest has to delineate landtypes based on morphology, parent material, soils, and vegetative types. Some calibration in WATSED is possible by including correlation factors for surface erosion curves, sediment delivery ratios, and mass acceleration. Other data required are: a ***primary general watershed database*** that includes total watershed area, estimates of total natural sediment delivery, precipitation, runoff (acre feet) and water yield (by elevation) and a ***secondary watershed database*** that stratifies the watershed by landtype, precipitation zone, and acres. The **a*ctivities database*** catalogs activities within the watershed, stratified by landtype—both past and proposed. The activities include roads, logging, site preparation, grazing, mining, and fire. All the necessary databases are indexed to specific watersheds and landowners via unique Water Resource Council Codes.

WATSED models the following processes:

- water yield
- equivalent clearcut acres (calculated under assumed 'recovery' rates)
- road prism disturbance
- area disturbed by mining, grazing, and other activities
- runoff increase caused by activities
- runoff distribution
- natural sediment yield
- management-induced sediment yield (surface and mass)
- total sediment delivery from logging, roads, mines, and grazing
- sediment routing to critical reach
- mitigation effectiveness

WATSED has improved model efficiency in several areas. The following is a list of those improvements.

- allows for local calibration and validation of databases and coefficients
- allows for linkages with other land ownerships
- allows for consideration of grazing, mining, and other activities
- erosion curves and sediment delivery ratios have been updated; local calibration is required
- mass erosion effects are considered out to 20+ years
- vegetative and hydrologic recovery allows for local calibration
- the model can analyze up to 4 watersheds at a time
- provides for continuity and consistency across Northern Region Forests
- computer capability has been enhanced

WATSED still has some weaknesses. The following is a list of those shortcomings and areas that need improvement.

- erosion curves have not been developed for grazing, mining, and other activities; area specialists must develop these relationships
- erosion curves for additional geologies need to be developed—only granitics and nongranitics are currently modeled
- surface erosion curves extend only to a period of 7 years after activity initiation
- mitigation is treated as a flat reduction, linear function
- the model analyzes watersheds up to a maximum size of 50 square miles (usually 3rd order)
- routing of sediment function is still primitive
- the sediment accumulation function has not been validated and should only be used with on-the-ground data on instream sediment conditions
- the model has not undergone a "hard" peer review in the scientific community
- only provides estimates of average sediment delivery; does not have capability to model specific events or annual conditions.

WATSED, although shown to be a useful tool for estimating sediment delivery and degree of watershed impact, is a model having the weaknesses typical of models in general and, as such, should be used with a healthy dose of caution and awareness. Outputs and estimates still require professional interpretation by a resource specialist who is familiar with the local watershed being modeled and the actual instream conditions of the channel. A **"factor of safety"** should always be employed when using this model. Validation of the sediment model is needed at the 2nd-3rd order watershed level; extending validation to the 4th-5th order watershed (salmon-bearing) should follow this.

Riffle stability index (RSI)

The **Riffle Stability Index (RSI)** is another method that attempts to deal with dynamic equilibrium of watersheds and channels (Kappesser 1993). It is aimed at quantitatively estimating mobility of channel substrate under estimated streamflow magnitudes of a given recurrence interval. Moreover, the RSI may help assess cumulative effects on the stability of substrate. The channel substrate conditions integrate all processes in the watershed including alterations caused by past and present activities in the watershed above the point of measurement. Furthermore, it can be linked to hillslope processes and changes in fish habitat components (Kappesser 1993).

The procedure assumes that riffles are the logical places to evaluate bed material mobility because they are "...remnant channel features formed at higher flows and are major storage locations of bed material" (Beschta and Platts 1986). The method assumes that the relative mobility of bed material on the riffle at estimated bankfull discharge can become an indicator of the riffle's equilibrium, aggradation, or degradation (Kappesser 1993). In essence, the method is akin to the residual pool volume method (Lisle and Hilton 1992) which measures fine sediment volumes in pools as a measure of sedimentation and channel stability. If the largest mobile particle sizes can be identified within the distribution of sizes present, and if the percentile of the distribution can be determined, then this can be used as an index of bed material mobility and "dynamic equilibrium" (Kappesser 1993). Kappesser (1993) has defined this relationship as the Riffle Stability Index. The higher the levels of fine sediment in riffles the more mobile is the riffle under expected streamflow magnitudes and the higher the RSI values. Thus, high RSI values are taken to be an indicator of both high levels of sedimentation and mobile stream substrate that is likely to result in increased degradation such as pool in-filling.

Only the major steps of the RSI method (Kappesser 1993) will be summarized in this description. A network of channels within a system is stratified into homogeneous reaches according to the Rosgen (1985 and 1994) classification scheme. Three representative riffles are sampled within a Rosgen reach. Three transects per riffle are measured for particle size distribution using the Wolman pebble count procedure (Wolman 1954). These data are used to estimate cumulative particle size distribution in riffles (see Bauer and Burton 1993). The next step involves identifying a "fresh" depositional feature such as a point or central bar and then measuring 10 to 30 of the largest dominant particles residing on the "fresh" bar. From these data, the geometric mean particle size can be calculated for the largest particles. This size is then used to estimate the largest common size of bedload transported in the channel at channel forming flows. In some streams, "fresh" depositional areas may not be available and a calculation of tractive forces is substituted. In these cases the channel cross-section and the water surface gradient at bankfull discharge have to be determined by standard surveying techniques.

Ten cross-sections (transects) per dominant channel type per reach should be sampled for surficial sediment. Water yield and flow characteristics can be assessed with the WATSED model and calibrated with data from permanent discharge measuring stations (stage recorders). Establishment of relatively inexpensive, permanent stations for obtaining continuous discharge records to augment USGS data is recommended over taking periodic field measurements of discharge because of the need for long-term, accurate flow information. These data provide a basis for interpreting temporal changes in substrate, pool distribution and morphology, depth and frequency of scour/fill, LWD transport, sediment deposition and transport, summertime water temperature maxima and diel variation. Also, because most USGS gages are located on relatively large rivers with differing hydrologic properties (gradient, temperature, precipitation type, etc.), extrapolation to smaller salmon-bearing watersheds can be more accurately accomplished with expansion of the gage network.

Data interpretation involves the comparison of the geometric mean size of the dominant large particles obtained from the "fresh" bar to the cumulative particle size distribution of riffle material. The RSI is the percentile value from the cumulative particle size distribution in the riffles, specifying the particle size that is equal to the geometric mean diameter of dominant large particles sampled from the fresh deposit. For example, if the geometric mean of dominant large particles was measured at 43 mm and a particle of 43 mm diameter corresponded to the 65th percentile in the full size distribution of particles in the riffle, then the RSI would be 65. It is the size class percentile that becomes the index (Kappesser 1993). According to Kappesser (1993), the index numbers can range from 50 to 100. An index number of 50 is considered to represent a stable riffle in an alluvial channel. Conversely, an index value of 100 represents a riffle that is totally aggraded. The range of indices from 50 to 99 is considered to represent a continuum of aggradation with no single number defined as a threshold. However, logical groupings can be made for general watershed interpretations. Index numbers less than 70 suggest systems that are in dynamic equilibrium whereas, numbers greater than 90 indicate that the watersheds are out of equilibrium or have exceeded the geomorphic threshold. Numbers that fall in the "gray area" (70-90) require the interpretations of professional water specialists. However, these bounds have not be verified; deleterious channel change may occur at RSI values below these estimated thresholds. Consequently, the model requires validation before it can be relied upon.

The RSI model is best applied to gravel and cobble streams in single channels. It cannot be directly applied to sand bed channels whose bedforms are ripples and dunes. The procedure has been extensively used on the Panhandle NF in 'belt' geologies. Although the data are somewhat limited, the model appears applicable to other areas and geologies such as granitics and basalts (monitoring report, Clearwater NF 1993). A statistically significant relationship between RSI indices and residual pool volumes has been obtained on the Panhandle NF for several key fisheries drainages (D. Cross, pers. comm.). However, we must caution that the RSI does not appear to be a sensitive measure that can act as an early warning tool. It appears to document what is already obvious about the level of degradation in salmon habitat and the amount of watershed development that resulted in worsening RSI condition.

Riparian Condition

A two stage approach is recommended for monitoring riparian areas to determine potential sources of large woody debris, bank stability, and the sanctity of the riparian reserves. The first stage will include low elevation (<3,000 feet) aerial photography of all riparian areas within a test stream and its control analogue. Grant (1988) has successfully developed an aerial photographic technique (RAPID) to evaluate the effects of riparian canopy disturbances on channel conditions over time. The RAPID method looks at changes in width of the riparian canopy to indicate major disturbances (floods, landslides, debris flows) to the channel system. The RAPID technique requires photographs in the scale range of 1:12,000 to 1:15,840. Larger scale photographs are recommended because individual features are easier to identify (Grant 1988). Our primary objective in use of such a method is to detect (early) baseline conditions and impacts on the riparian reserve violating management prescriptions. Over time, it may provide useful supplementary information on cumulative effects on watershed and channel stability (Grant 1988).

Monitoring flights (randomly selected days) should be conducted at least tri-annually prior to, during, and after periods of development activities. Field confirmation of riparian conditions should be conducted at least once a year in representative channel types of the upper, middle, and lower reaches of the stream. Field surveys should include measurements of buffer strip integrity and bank stability (visual estimates calibrated against "true" measures). Monitoring of control streams will focus on the impacts of natural events such as fire and blow-down and will not require field validation unless problem areas show-up in aerial photographs.

The percentage of banks that are actively eroding (conversely, percentage bank stability) is a useful index of channel stability. It could be related to degree of natural or management related disturbances to the riparian zone. This index has been found to be related to the percentage of drainage area logged in south coastal Oregon streams (Frissell 1992).

In-Channel Monitoring

Water Chemistry

A comprehensive water chemistry survey of the watershed system is necessary to establish background profiles of chemical constituents. This effort is a requisite complement to the other basic reconnaissance monitoring of Level 1.

Samples of water quality need to be obtained by widespread geographic coverage within stream systems from headwater tributaries to the mainstem channel. MacDonald et al. (1991) have described standard considerations and procedures for sample procurement and sampling effort. Samples should be depth-integrated (e.g., suspended sediment) and correlated with major stages of the hydrograph such as base flow, ascending limb, peak flow, descending limb, and base summer and winter flows. Samplers that collect samples more frequently during major changes in discharge can capture information on rate of change in chemical constituents with flow dynamics. Samples should then be sent to a certified laboratory for processing.

A core set of water chemistry parameters should be measured: **pH, conductivity, nitrogen, phosphorus, heavy metals, herbicides, pesticides, petroleum products, major cations and anions.** Mining effluents and toxic spills will present some special considerations in water quality monitoring. Panther Creek, a tributary of the Salmon River, is a well-documented case of chronic pollution from a mining project that to this date has not been totally eliminated. The extensiveness of mining activity in the Northwest presents many serious water quality monitoring challenges. The presence of mines require attention to heavy metal pollution, shifts in pH, cyanide, and diesel fuel contamination, in addition to increased sediment delivery from tailings and road construction and traffic. Dissolved oxygen and pH should be determined in the field by ***Standard Methods*** (see MacDonald et al. 1991). It is not normally necessary to sample water quality on an annual basis unless there are mining, grazing (feedlots, heavy concentrated grazing), herbicide, pesticide, or fertilization projects within the system or if representative recovery trends are being studied. Chemical treatment (usually magnesium chloride or sulfate) of riparian road surfaces for erosion control may require periodic sampling before, during, and after treatment. Riparian roads that are used for transportation of toxic materials should have their adjacent channels sampled on an annual basis especially during base flows. Unless special circumstances warrant, trend sampling of water quality can be conducted at a 3-year interval.

Fish Habitat and Population Reconnaissance

Beyond the watershed considerations, a comprehensive inventory of both fish habitat and populations must be conducted to establish baseline conditions and trends, and identify salmon habitats. A basin-wide or system approach is required to delineate the extent and intensity of habitat utilization within the system and to locate the ***"critical"*** spawning and rearing habitats. The more intensive Levels 2 and 3 of monitoring are dependent upon this assessment. The primary objectives are to cover the system and identify where the salmon habitat is located plus its extent and quality.

Although there are other methodologies that can be employed, the Hankin and Reeves (1988) procedure has been widely used throughout the Columbia River Basin. The methodology is based on research conducted by Hankin (1984) in which he constructed the statistical foundation for estimation of fish abundance and total habitat area in small streams. Basically, the procedure involves correlations between visual estimates and corresponding ***"true"*** habitat areas and fish numbers (i.e., development of calibration factors). This procedure is a useful means of estimating metrics for habitat components and fish populations. It is especially useful in developing basinwide estimates for CSP parameters such as percentage surface fine sediment, cobble embeddedness, percentage bank stability. Spatial distribution of LWD has been found to be so clumped that it may be better estimated as total counts (Overton et al. 1993). Pool areas and channel widths could be estimated using the Hankin and Reeves approach, but pool depths or volumes may likely be more accurately estimated from measured depth transects for primary pools (i.e., those exceeding 1 m in depth) by measuring all such pools in the basin or by sampling every *n*th unit.

Sampling designs assume that habitat units are first stratified according to channel unit (e.g., pools, riffles, glides, alcoves) and location (e.g., upper, middle, lower reaches) so as to generate a finite set of channel unit/location strata. Primary units within strata are of unequal sizes, equivalent to the natural habitat units, and each habitat type/location stratum is sampled independently. Because of independent sampling, estimates of fish abundance and habitat area can be added within strata to give estimates of total fish abundance or habitat area for specific habitat types of the entire stream (Hankin 1984; Hankin and Reeves 1988).

The procedure is described in detail by Hankin and Reeves (1988). Only the major components will be described here. Concerning total habitat, visual estimates of habitat areas are made for all habitat units within a given habitat type/location stratum. A systematic sample of units (every 1 in *n*) is drawn from this stratum and accurately measured for each unit appearing in the sample based on uniformly spaced measurements of stream width. The assumption is that these accurate measures of habitat area have negligible errors so that they can be equated with the ***"true"*** habitat area. Visual estimates are then calibrated with the "***true***" measures according to the equations presented in Hankin and Reeves (1988).

In field testing of their procedure, Hankin and Reeves (1988) reported that visual methods were extremely effective in estimation of habitat areas. They reported estimates with narrow 95% confidence bands (±13-16% of estimated total areas of pools and riffles). They further stated that survey results allowed the construction of detailed maps of the locations and sizes of all habitat units. These characteristics have enormous utility for monitoring purposes as they are suited for detecting changes in habitat profiles within stream strata that could be affected by development activities. Moreover, the data are suited for treatment and synthesis in a ***Geographic Information System (GIS).***

In estimating fish abundance, a systematic sample of ***n*** units is selected from a given channel unit/location stratum and paired independent diver counts of fish numbers are made for each unit in the sample. In a random subsample of these selected units, a more accurate method (usually electrofishing) is used to allow calibration or adjustment of diver counts and calculation of errors of estimation within selected units. Because of the endangered status of Snake River salmon, electrofishing is not recommended because of risk to salmon. Therefore, visual estimates (snorkel diving) of salmon may have to stand alone without calibration to "true" measures. However, in many streams that are deep or have a high degree of habitat complexity, snorkel diving will be the most accurate method of estimating fish abundance. Efficiency of sampling by electrofishing varies by age class (House 1995), species, and channel unit sampled (Peterson and Rabeni 1995). Reported variation between visual estimates of abundance by independent divers has been small (Northcote and Wilkie 1963; Schill and Griffith 1984). The area-density method described by Everhart and Youngs (1988) could be used in lieu of electrofishing to estimate fish abundance.

Mapping and spatial analyses of habitat and fish population densities are feasible using GIS technology. Level 1 monitoring provides the necessary data layer for comprehensive coverage of the stream system, identification of ***"critical"*** spawning and rearing areas, fish distribution and abundance, and initial identification of limiting factors. Such mapping is possible using either the Hankin and Reeves systematic sampling procedure or a total inventory procedure.

Hankin and Reeves methodology is not recommended without some caveats. The procedure requires highly trained and experienced observers who are capable of consistency and accurate bookkeeping (Hankin and Reeves 1988). Inexperienced surveyors and divers usually have problems with consistent identification of habitat types and fish species/age classes, respectively. Most users of the procedure in the Basin have modified it to include more measured parameters. Their objectives have been to facilitate consistency and repeatability. In addition, Overton et al. (1993) reported that for certain habitat variables, such as LWD, a highly clumped spatial distribution makes it necessary conduct a total inventory.

Sediment Transport

Sediment monitoring "close to the action" can be conducted at the point of origin (project monitoring) by measuring suspended or bedload sediment transport from small tributaries or road ditches draining project sites. For estimates of sediment transport to be meaningful, sediment concentrations must be monitored on a continuous basis during and after the main project activity to assess sediment delivery during peak flow events or saturated soil/road surface conditions. By utilizing water discharge, suspended sediment concentration, and bedload data collected during rising and descending portions of the hydrograph, total sediment transport can be calculated. As a relative measure of sediment delivery from a project, sediment concentrations from upstream to downstream of the project can be useful as an effectiveness monitoring technique.

Another form of sediment monitoring at the sources includes assessing the baseline erosional status of the entire watershed system. This accounts for changes in stability that have arisen for multiple current and past projects plus natural conditions on the landscape. The exposure of numerous sites with varying land management activity histories to series of climatic conditions establishes an erosion/sediment delivery baseline. In conjunction with the instream LWD, channel morphology, effective gradient, riparian vegetation density, and stream power (all interdependent characteristics of the stream network), the ability of the stream to store/transport sediment is determined. Reconnaissance of watershed condition, road density and location (i.e., landtypes occupied), mapping of sites of surface erosion and mass failures (current and potential), and areas involved in grazing and mining identify a baseline for sediment yield.

Given varying levels of development of a watershed of a certain type, sediment delivery to the stream system and transport operate at various equilibrium points. Major shifts in the calibration are expected as a watershed makes a transition from damaged to recovered. Suspended and bedload sediment monitoring are effective tools for establishing discharge versus sediment transport calibrations in relation to general level of watershed disturbance (Rosgen 1980, Ketcheson 1986). These calibration curves identify current watershed equilibrium points between delivery and transport and provide a technique for monitoring watershed-wide recovery.

Substrate Fine Sediment

Fine sediment deposition reduces fish production directly by altering spawning gravel composition and reducing rearing habitat diversity by infilling of interstitial spaces in coarse substrate. Loss of interstitial space reduces growing season hiding and escape cover and also overwinter refuge sites. Alteration of spawning gravel composition by fines reduces intragravel water flow velocity, intragravel dissolved oxygen, crushes incubating eggs and alevins, and provides impenetrable barriers to successful emergence. These biological effects form the proper focus for field monitoring (i.e., suggest the parameters, monitoring sites, and monitoring times).

Ocular estimates of surface substrate composition have been employed successfully (Platts et al. 1983, Petrosky and Holubetz 1986, as cited in Torquemada and Platts 1988). Torquemada and Platts (1988) recommend the use of ocular estimates as a rapid, low cost inventory technique suitable for extensive analysis. This is an appropriate tool for use in a coarse screen procedure. Torquemada and Platts (1988), however, caution that ocular estimates introduce observer bias. This is nothing new in sampling; all techniques are affected by degrees of observer bias. The Hankin and Reeves procedure for establishing calibration factors to a method believed to be more accurate can be performed for surface fines just as for other parameters such as length and area estimates. In the Blue Mountains of Oregon the ocular estimate of surface fines had a resolution of ± 5% relative to estimates made using the direct measurement method of Burton et al. (1991)(Rhodes, CRITFC hydrologist, unpubl. field notes 1994).

Not all types of sediment monitoring need to be performed in every sampling site. However, for purposes of verifying the adequacy of the CSP, it is necessary to employ multiple methods for monitoring each parameter. For example, fine sediment should be monitored at depth to determine whether estimates of surficial fine sediment accurately reflect fine sediment at depth. Concern over the validity of any single fine sediment monitoring parameter can be illustrated from data on the South Fork Salmon River. The Poverty spawning site has a mean fine sediment level of 31.2% (Newberry, 1992, as cited in NMFS (1995) draft biological opinion for the Thunderbolt salvage sale). Using the USFS (1983) model (also cited by NMFS), one would expect an egg-to-emergent fry survival rate of approximately 33% at this fine sediment level. However, NMFS indicated that the USFS measured actual survival rates of 1.4% at the Poverty spawning site. This extremely low survival appears to be attributable to a high concentration of fine sediment at depths in the substrate where eggs are deposited relative to the percentage of fines in entire substrate cores (the conventional method applied to monitoring fines under the technical guidelines). For biological monitoring it is important to employ multiple methods to calibrate the more commonly used methods. For example, Hankin and Reeves (1988) use electrofishing to calibrate visual estimates. In addition, population trends can be effectively supplemented with estimates of trends in fecundity, survival, food availability, level of competition, etc. These additional biological parameters give insight to population trends.

Various authors have used varying spatial sampling techniques for measuring substrate conditions. Torquemada and Platts (1988) estimated substrate sediment parameters for 100-m reaches using 33 transects spaced at 3-m intervals. Each transect was sampled at 3 locations. Grost et al. (1991a) sampled sediment on 5 transects, taking 3 samples spaced 0.5 m apart on each transect. Sample points selected had similar depth, velocity, and substrate. Hogan (1987) characterized substrate at transects spaced 10 m apart. Skille and King (1988) in embeddedness monitoring recommend use of 10 transects spaced at 2 times mean channel width. The reach length sampled is then 20 times mean channel width.

Considering the variety of methods that have been effectively used in the literature for substrate composition, various methods of site selection can be proposed. For area-or volume-based methods, sampling strategy options include:

- estimate condition by channel type and channel unit; employ ocular sampling method to provide estimate for total unit. Select a channel type(s); stratify channel units within each channel type; make ocular estimates of surface sediment composition by particle size class and calculate geometric mean and standard deviation for each channel unit by integrating the characteristics of the entire unit. If the unit is not homogeneous in composition, subdivide into relatively homogeneous units on the basis of surface texture so that composition can be integrated. Estimate channel unit area and percentage of the unit comprised by any subunits (i.e., differentiated by substrate distribution).

- criteria: same as above; employ transect method of sampling. Select a channel type(s); stratify channel type(s) by channel unit(s); select a reach with length of 20 times mean stream width or 100 m, whichever is greater. Establish transects at intervals of 2 times mean stream width and sample substrate composition (ocular and quantitative methods, such as shovel, freeze core, pebble count) at 0.25, 0.5, and 0.75 x stream width. Record depth, velocity, and channel unit type for each point.

- use statistical considerations of the Hankin and Reeves sampling approach for sampling channel units if the channel unit distribution is complex (i.e., short channel units that alternate longitudinally and laterally). Take 3 random samples per channel unit using a quantitative method (shovel or McNeil core, see Grost et al. 1991a) and at least 21 samples total per channel unit type in the reach.

The CSP (Rhodes et al. 1994) document recommends assessment of surface fine sediment (%) in historic spawning habitat as an index to the quality of the egg incubation environment for salmonids. Torquemada and Platts (1988) demonstrated a high correlation ($P < 0.01$) between ocular estimates of percentage fine sediment and those made by the surface transect method of Platts et al. (1983). The ocular method was the most rapid and cheapest method and is conducive to making watershed-wide surveys. They recommended use of other methods of measuring substrate fine sediment, such as measured cobble embeddedness (Kelly and Dettman 1980, Burns and Edwards 1985, as cited by Torquemada and Platts 1988), in more intensive work. The method of Platts et al. (1983), while based on measurement of lengths along a transect occupied by material of various particle size classes, involves reporting only dominant substrate for each 0.3 m along the transect lines. This procedure seems likely to underestimate the percentage of subdominant particles. Platts et al. (1983) assumed that biases involved in assigning a dominant size class for each segment of the transect would be compensated by large sample size. This assumption does not appear to be warranted because fine sediment may need to comprise greater than 50% of each linear segment before being classified as dominant. The ocular method for estimating surface substrate composition evaluated by Torquemada and Platts (1988) was based upon data taken from 0.3 m-sample frames (0.09 m^2 area), which the authors considered to sample too small a percentage of the channel. To compensate for potential weaknesses in each method, we recommend that surface fines be estimated by employing the method of Platts et al. (1983), but instead of recording dominant substrate, record percentages of each substrate class for each 1 m along the transect (see Overton et al. 1993). In addition, this method should be correlated with ocular estimates for 1 m^2-samples cited on transects, such as those recommended by Torquemada and Platts (1988). The larger sample sizes (1m^2 rather than 0.09 m^2) are well within observer capability to visualize, will better capture presence of large size substrate material, and provide a larger sample of the channel for a given sample size. Data on percentage of bed material within various particle sizes can yield biologically useful indices, such as geometric mean and standard deviation of the entire bed composition as well as percentage <0.85 mm or 8 mm, thereby providing estimates of percentage fines as well as total bed composition (MacDonald et al. 1991). Bevenger and King (1995) recommend the use of 8 mm as the cutoff size separating fine from large particles for detecting the impact of land management in altering particle size distributions.

The Wolman pebble count technique is a quantitative technique that has been commonly used to provide an index of surface particle size distribution, Bevenger and King 1995). It may be biased against smaller size fractions (MacDonald et al. 1991) but can provide statistically reliable estimates of pebble size and larger fractions. The pebble count method can be used to define the cumulative frequency distribution of particles sizes, thereby identifying the D_{50} particle size (Bauer and Burton 1993). Despite these cautions about potential biases, it is a technique that is commonly used to validate the reliability of ocular estimates of particle size distributions (Overton et al. 1993).

Another useful approach to estimation of surface fine sediment composition, especially in well mixed substrate material, may be using a grid method. By placing a sample frame (22 inch frame with 2 inch squares) at points on each transect, determine the percentage of all grid intersections with fine sediment less than 6.4 mm (Bauer and Burton 1993). Because of lateral variation in surface particle composition, several grid frames may be necessary to describe the entire wetted streambed.

Subsurface sediment composition is critically important in its effects on fish egg development, survival, and fry emergence. For surface fine sediment to be a monitoring tool for extensive surveys, it is important to investigate and establish a correlation between surface and subsurface fines. Streambed substrate material is frequently reported to be stratified with a topmost pavement layer, a subpavement, and then a third layer of deep sediment (Everest et al. 1981, Diplas and Fripp 1991). The thickness of each of the top two layers is roughly comparable to that of the largest particles found in each layer (Diplas and Fripp 1991). The subpavement layer contains a higher percentage of fines than either the pavement or deep bed layers and this deposit is not influenced by the sediment concentration in flows as is the surface pavement layer (Diplas 1991), although Carling (1984) reported infiltration of fines into the subpavement to be substantial even at low sediment concentration. The depth of penetration of fine particles into the bed is mediated by the size difference between the coarse particle matrix and the infiltrating grains (Lisle 1989, Lisle and Eads 1991, Diplas 1991) and also is a function of the boundary Shields stress. The finer the matrix, the less and the shallower the depth of penetration. Diplas (1991) reported the depth of a sediment seal to be at a distance of $3D_{90}$ below the surface. Lisle (1989) found that when the ratio of particle diameters of the substrate framework particle to intruding particle is >60, intrusion can occur to the base of the bed. Removal of fines from the subpavement is dependent on bed mobilization (Diplas 1991, Scrivener and Brownlee 1989). The differentiation of pavement and subpavement caused by the different response times to magnitude of flow could make single estimates of surface fines unrepresentative of subsurface fines (Platts et al. 1983, Platts et al. 1989). However, surface fines do influence suspended sediment concentrations and transport and thus strongly control subsurface levels. For example, during low flow with very little surface fines there is little transport and little intrusion. During low flow with high surface fines there is high transport and high intrusion. Lisle (1989) found that fine bedload, not suspended sediment, was the main component of infiltrating fine sediment. After spawning females clean gravels in the redd, it is common for the combination of infiltration and scour and fill to deposit fine sediment at various depths in the active bed (Lisle 1989). Surface-based assessments generally are overestimates of the percentage of coarse particles by volume for the top two layers (Everest et al. 1981, Diplas and Fripp 1991, King and Thurow 1991, Carson and Griffiths 1987, Scrivener and Brownlee 1989), making surface fines by area apt to be a minimum estimate of the fine sediment problem. However, Lisle (1989) found that surface material in spawning gravels of three northern coastal California salmon-bearing streams was slightly finer than the subsurface material.

After manual removal of fines <6.35 mm in four Idaho sampling sites, King and Thurow (1991) observed intrusion of fines into cleaned gravel during low flow periods (September and

October). The percentage fines <6.35 mm and 0.85 mm, respectively, continued to increase from October to March. Grost et al. (1991b) reported that fine sediment <0.85 mm infilled brown trout egg pockets throughout the winter period, making egg pocket substrate composition indistinguishable from that of the tailspill material. This evidence is counter to the idea that once a female cleans a redd and deposits eggs in the egg pocket that the egg pocket will not be subject to fine sediment intrusion during fall or winter incubation when the redd is located in a degraded channel (see Everest et al. 1987). Intrusion of fine sediment into the streambed is probably more commonly viewed as unavoidable where fine sediment is in transport (Lisle 1989); transport is generally high when percentage surface fines are high. It appears that variation in depth of egg deposition combined with uncertainty about whether a sediment seal would form, shielding the egg pocket from further sediment infiltration, and the unlikely event that subsurface sediment composition can remain higher quality than surface conditions make it dangerous to ignore adverse surface fine sediment conditions.

Everest et al. (1982) criticized the common technique employed in fisheries surveys of estimating surface fine sediment by area. From freeze core sampling Everest et al. (1982) had noted greater concentrations of fines in the 20-30 cm depth zone (where chinook egg deposition would occur) than at the surface. In fact, the percentage sand at the surface versus the 20-30 cm zone was reported at a site to be 10% versus >40%, respectively. King and Thurow (1991) reported that the concentration of fines in the 0-10 cm layer was not significantly different from that in the 10-20 cm layer, but the 20-30 cm layer had a significantly greater percentage of fines. Grost et al. (1991a) noted that the lower half of freeze core samples from small Wyoming streams had a significantly greater percentage of fine sediment that the upper half. Bands of high levels of fine sediment in subsurface deposits could easily act as impenetrable barriers to fry emergence despite relatively benign surface fine sediment conditions. Depth of spawning among salmonids varies by species, with smaller species depositing eggs nearer the surface (Bjornn and Reiser 1991, Nawa et al. 1988) where they may be subjected both to increased frequency of bed scour or mobilization and fine sediment infiltration into the subpavement (Scrivener and Brownlee 1989). Percentage surface fine sediment likely provides an underestimate of the magnitude of fine sediment at depth in spawning areas. Occasionally, the subsurface fine sediment conditions are far worse than can be inferred from surface conditions (Everest et al. 1982). However, ease of measurement over extensive stream channel area makes it a useful screening tool and known correlations to subsurface conditions for individual streams or reaches may make it a more sensitive measure. The surface sediment standard in the CSP is not predicated on any notion that it is a perfect predictor of the subsurface. Rather it is predicated on data showing survival is a function of surface fines and subsurface through their control on water flow and emergence; surface fines are proportional to subsurface fines; trends in surface fines reflect subsurface trends; and surface fines are fairly responsive to the existing sediment delivery and loads in transport and are easy to monitor.

Grost et al. (1991a) recommended the use of excavated cores or shovel samples to measure fine sediment levels to a predetermined depth in the streambed rather than the more laborious freeze core method (e.g., see Everest et al. 1982). In fact, they determined that the excavated core and shovel methods performed better than the freeze core method in representing known test substrate material. The freeze core method, however, is useful in cases where stratification of substrate material by depth must be known. Freeze cores take such a small portion of substrate that intersample variation is high, requiring a large number of samples to characterize the substrate (Lisle and Eads 1991). The shovel sample is the most convenient method for estimating fine sediment composition of substrate in remote locations because of lightness, but a McNeil core provides a means to excavate substrate in fast water without loss of fine sediment. Lisle and Eads (1991) recommend use of a 30-cm diameter McNeil core implanted to a depth of 30 cm when taking bulk samples and concluding sampling by thoroughly agitating the water inside the coring cylinder to estimate remaining fine sediment concentration.

In addition to issues of substrate stratification and infiltration by fines, there is debate about the survival benefit attributable to conditions in the egg pocket versus the redd or spawning gravel. This stems from a report by Chapman and McLeod (1987) indicating that Platts had measured a difference in D_g between egg pocket and surrounding material. Grost et al. (1991b) found that substrate composition in brown trout egg pockets was not significantly different from tailspill material after winter incubation, although they did differ initially after egg deposition. King and Thurow (1990)(as cited by King and Thurow 1991) reported that steelhead egg pocket material was significantly different from surrounding redd material but King and Thurow (1991) could not discern a difference between natural redds, artificial redds, and surrounding spawning gravel. They were unable to adequately distinguish egg pockets within the redds.

Young et al. (1991) reviewed 15 measures of substrate composition on their ability to predict STE for cutthroat trout. Their review revealed the large number of particle diameters that have been used as predictors of STE in the literature, ranging from 0.75 mm to 6.4 mm. In addition, the distribution of particle sizes has been described by various statistics such as arithmetic and geometric means, median, fredle index, skewness, etc. Young et al. (1991) found D_g (geometric mean particle size) to be the best predictor of STE in a comparison of D_g, F_i (fredle index), F_m (modified fredle index), and percentage fines <6.3 mm. They noted that although STE prediction is better when done using D_g (though not greatly better), the percentage of particles <0.85 mm in channel substrate were directly linked to land uses such as road building (see Beschta 1982, as cited by Young et al. 1991). As a result Young et al. (1991) recommend use of at least both D_g and percentage fines <0.85 mm to monitor changes in substrate composition and to predict effects. Of course, the size fraction being affected by land use may vary depending on the particular geologic and land use effects. Streamside mass failures may result in delivery of some coarse particles to the channel whereas surface erosion from logging and road surfaces would contribute primarily finer sediment.

Laboratory studies of STE versus gravel composition are often criticized because mixtures of gravel in the laboratory are frequently simple and do not represent the wider diversity of particle sizes found in the field (Everest et al. 1982). Young et al. (1991) reported that when two particle size distributions were skewed and uniform, respectively, and both had 25% fines <3.35 mm, STE for cutthroat trout was 39% and 11%, respectively. Although skewness of substrate particle composition can create variation in STE observed under experimental conditions, this phenomenon does not discount the importance of minimizing addition of fine sediment to streambeds. Despite occurrences of higher than expected STE under laboratory conditions with high percentages of fine sediment and skewed particle distributions, there are numerous uncertainties associated with field conditions that are likely to make it unwise to downplay the significance of elevated fine sediment levels. There is uncertainty regarding the natural level of variation in skewness of particle distributions; the tendency of substrate having greater proportions of coarse sediment to cause greater rates of fine particle sedimentation in interstices; the tendency of substrate having an initially large fraction of coarse particles to develop fine sediment caps. For these reasons it should not be assumed that variations in skewness will compensate for high percentages of fine sediment.

Despite different opinions about the most effective indices for describing substrate composition, predicting effects on STE, or sensitively monitoring land use impacts to substrate composition, it is clear that increased levels of fine sediment in substrate from watershed development negatively affects salmonids so percentage fines and some measure of the full particle size distribution should be determined as a minimum in trend monitoring, as recommended by Young et al. (1991).

Survival-to-emergence of eggs deposited in the egg pocket is a function of apparent velocity, gravel permeability, dissolved oxygen, and presence of entombing fine sediment that may fill all voids down to a sediment seal or may actually fill the egg pocket itself (Scrivener and Brownlee 1989). All these parameters are interrelated and dependent on percentage fine sediment and overall composition of the substrate. It seems likely that percentage STE for any spawning reach is more strongly influenced by the overall substrate composition, percentage fines in the substrate to the depth of egg deposition, and percentage surface fines than by finer spatial scale differences in percentage fines between egg pockets and surrounding redd material. Rate of water flow through the entire subsurface gravel matrix seems likely to control flow through egg pockets contained in the matrix. Also, surface fine sediment must have a role in controlling the entry of water into the substrate at hydraulic control points (Scrivener and Brownlee 1989). It seems likely that these factors influence STE more than potential differences in gravel composition between egg pockets and redd material. Even if a difference does initially exist between the egg pocket and the surrounding material, it has been noted here and in Rhodes et al. (1994) that high ambient sediment levels can result in rapid infiltration. For these reasons, surface fine sediment is an appropriate metric for rapid assessment of the overall sediment regime. This metric is responsive to land use, is linked to survival of eggs and juveniles, and is an indicator of related hydrologic and geomorphic behavior of the stream having implications for fish survival.

Intra-Gravel Dissolved Oxygen

The use of the intragravel dissolved oxygen (IGDO) technique has been suggested as a surrogate for measuring fine sediment levels in the substrate. It can be either a direct or indirect measure for estimating the effect of fine sediment on incubating eggs. As a direct measure of the incubation environment, IGDO must reflect oxygen concentration within the egg pocket. This oxygen level is a product of ambient levels in the water column, oxygen demand in the substrate, inflow of de-oxygenated groundwater, and oxygen depression caused by egg respiration in egg pockets. If the DO probe misses the egg pocket, the DO level measured will not be one that reflects egg respiration effects. If DO levels in redd material are marginal because of a high BOD associated with fine organic matter, conditions in the egg pocket can be even worse (Maret et al. 1993). So, if a large percentage of DO probes miss the egg pocket, IGDO measurements will underestimate the direct effects of oxygen depletion on egg respiration. IGDO is also apt not to be a reliable indicator of indirect effects of fine sediment. That is, the ability of fine sediment to entomb alevins in the gravel is a limitation to survival that may not be strongly correlated with IGDO. Scrivener and Brownlee (1989) found that entombment, not IGDO, was the most significant source of mortality in a British Columbia river producing chum and coho salmon.

Maret et al. (1993) assessed the use of IGDO on brown trout spawning habitat in Rock Creek, Idaho. They found that a mean percentage fine sediment (<2mm) of 9.0% and 16.2% in control and downstream managed reaches, respectively, corresponded to a mean STE of 48% and 17%, respectively. This study revealed that a small increase in mean percentage fines of spawning gravel results in a large reduction in STE and in percentage DO saturation. They also demonstrated that STE was very low for IGDO of 3 to 8 mg/l but increased greatly above 8 mg/l. They found a negative correlation between IGDO and percentage fines, although they suggested that other factors, such as BOD can reduce oxygen concentrations, thereby limiting survival.

If IGDO is to be considered as a useful adjunct to surface and subsurface fine sediment monitoring and feedback to land management, biologically appropriate standards must be adopted for endangered species protection to be meaningful. Given the information provided by Burton et al. (1991) and Maret et al. (1993) for response of brown trout in Rock Creek in south-central Idaho, Idaho adopted IGDO standards that were extremely unprotective of the beneficial use. The proposed criterion was that "nonpoint source activities shall not cause intragravel dissolved oxygen in spawning gravels to decline below a weekly average of 6 milligrams per liter" (see Burton et al. 1991). These authors reported conclusions of Sowden and Power (1985)(as cited by Burton et al. 1991) who stated that mean IGDO should be >8ppm in redds to ensure >50% STE. An IGDO of 6 ppm is clearly not supportive of salmonid survival. Burton et al. (1991) (their Figure 1) showed that IGDO was 6 ppm at 50% fines <6.3 mm. Alevin entombment at very high percentage fines levels such as this is not considered when only IGDO is measured. Maret et al. (1993) found that providing an IGDO of 6 ppm resulted in only 5% STE versus 50% STE at 10 ppm. In addition, they found that STE was only 17% when mean percentage fines (<2.0 mm) was only 16.2%, but improved to 48% at 9.0% fines.

Recommendation of a weekly <u>mean</u> IGDO of 6 ppm clearly is unprotective of salmonids. Such a standard allows minimum values that could occur daily or even once in a week-period that could be lethal within minutes. That is, there is no assurance that the IGDO would be 6 ± 0.5 ppm rather than 6 ± 3 ppm. This standard, in addition, makes no mention of percentage saturation or intragravel flow velocity, which can have a profound influence on STE. Maret et al. (1993) reported that STE improved from 10 to 48% as DO saturation changed from 70 to 90%. Maret et al. (1993) report negligible survival when mean IGDO was <8 ppm and 70% saturation. This study makes it clear that percentage fines (<2.0 mm) should be ≤9.0% so that IGDO would be >8 mg/l.

There appears to be no benefit to using IGDO in an attempt to reduce sample variance in predictions of effect of fine sediment. King and Thurow (1991) reported that the number of samples needed to assess percentage fines <6.35 mm at 5% precision and 90% confidence level was 5 redds. For DO 24 natural redds needed to be monitored.

Adoption of IGDO as a sediment monitoring tool seems to have few advantages over direct sediment monitoring and runs the risk of egg pocket disruption or underestimation of the level of impact, and still retains all the issues of sample variance that are present in substrate sediment sampling. In addition, requiring that all sampling be done in an egg pocket (1) may be impossible, provided that spawners are not present, (2) necessitates that only low gradient sites are monitored so that late warning of upstream impacts is the monitoring outcome, or (3) unfairly biases the samples to the few remaining sites where spawning adults sense sufficient upwelling in a gravel bed to make incubation feasible. Use of IGDO as an adjunct to surface (areal) and subsurface fine sediment sampling is desirable. For example, it can detect oxygen depletion caused by organic matter in spawning gravel (Ringler and Hall 1975, 1988), but these conditions placed on the IGDO tool make it a poor choice for rapid assessment of habitat of a listed species aimed at prevention of further damage.

Innumerable excuses have been given for years by federal land management agencies why they should not monitor fine sediment or why other variables are adequate surrogates. Efforts to supplant fine sediment monitoring with IGDO is an example of misguided attempts to avoid monitoring the direct cause of salmon mortality. See Appendix E for an explanation of the need for fine sediment monitoring and the nature of the obfuscation surrounding this variable.

Embeddedness

Embeddedness is a measure of the effect of fine sediment levels in reducing interstitial rearing area within the streambed. Fine sediment that surrounds cobbles, rubble, and boulders, filling interstices between these matrix particles, reduces the ability of juvenile fish of various sizes and life stages to use these crevice habitats for shelter. In addition, the creation of a fine sediment seal around large matrix particles and filling of voids reduces the total rock surface area for benthic macroinvertebrate production. As juvenile fish grow, their needs for escape cover during the day and for overwinter cover demand increasingly large interstitial spaces. Suitable alternatives to hiding in the substrate may not be available when fine sediment heavily embeds benthic habitat. The only other potential habitats reported to provide daily hiding cover or overwinter cover are root wads, deep overhanging banks, stable log jams, and deep pools with coarse substrate, although these are also frequently lacking in streams having impacts from logging, grazing, and other management activities (Hogan 1987, Ralph et al. 1994).

Torquemada and Platts (1988) compared three methods for estimating embeddedness: the hoop method (Kelley and Dettman 1980, as cited by Torquemada and Platts 1988); the modified embeddedness method; and the ocular method (modified from Platts et al. 1983). The hoop method has been used in Idaho successfully for many years and provides reliable, repeatable results (Burns 1984, Burns and Edwards 1985, as cited by Torquemada and Platts 1988). The hoop method requires that a circular hoop be tossed randomly within a pool tail-out area to define sample points meeting depth and velocity criteria. Cobble embeddedness (for particles 4.5-30 cm diameter) is measured as the height of a surface particle above the embedded plane divided by the total vertical height dimension of the particle in its original orientation. Burns and Edwards (1985) measured embeddedness on at least 100 cobbles located within the hoop. This method was modified by Torquemada and Platts (1988) to account for the percentage surface sand on the streambed, recognizing that when the bed surface becomes occupied by greater amounts of sand, the likelihood that some particles become fully embedded and thereby not recorded becomes greater. The higher percentages of sand present in the streambed result in underestimation of embeddedness, necessitating a correction for percentage fine sediment. Skille and King (1988) recommended that each hoop be treated as a unit and for embeddedness to be calculated as the mean of all particles within it meeting the size criteria. They also corrected for percentage surface area of fines, using the method of Torquemada and Platts (1988). Skille and King determined embeddedness on at least 30 hoops, but found that desired precision (5%) could be achieved with as few as 22 hoops.

The free matrix particle method, also reviewed by Torquemada and Platts (1988), is a fine sediment index that is inversely related to the embeddedness measure. It represents the tendency for pavement cobbles to be totally unembedded by fine sediment, or essentially sitting free on the surface, surrounded only by other similar particles. It is easier to assess than "measured embeddedness" using the hoop device.

Ocular embeddedness estimates using the method of Platts et al. (1983) rely on determining the percentage of the circumference of each cobble examined that is in contact with fine sediment, creating a sealing gasket. Ocular embeddedness estimates indicate something about the degree of difficulty that juvenile fish or macroinvertebrates would have in moving into the substrate below the circumference of a rock. This is a biologically meaningful measure. Even when cobbles are 100% embedded, as defined by this technique, there can be different percentages of the cobble exposed in the water current above the embedding plane, the principal feature defined by the measured embeddedness method. The measured embeddedness technique assesses the percentage of the vertical height of the cobble that protrudes above the embedding plane. The ocular and measured embeddedness methods may provide somewhat different results. For example, a cobble that is perhaps only 30% embedded, as determined by the ocular estimate of percentage of the circumference in contact with fine sediment, might have 50% of its vertical height below the embedding plane. The variable percentage of the cobble protruding above the bed can create some cover from current flow velocity, but cannot fully compensate for lost interstitial space. The measured embeddedness technique appears to be less subject to personal bias but may provide an embeddedness index that represents a combination of available interstitial space and vertical projection above the bed. These are both biologically significant outcomes of streambed sedimentation but with different implications.

Torquemada and Platts (1988) found that weighted hoop embeddedness (i.e., modified for percentage fine sediment) was significantly correlated to ocular embeddedness and to free matrix particle index. Continued validation of this correlation will be important so that the more rapid methods can be used for extensive, coarse scale monitoring (Level 1) and the more time consuming methods can be used at Level 2 or 3.

A technique that should be investigated for potential use as a coarse monitoring tool is ocular estimation of surface area distribution of particle sizes for channel unit types within Rosgen channel types. Geometric mean and standard deviation can be calculated from area estimates for 1-m^2 frames on transects within channel types. For each particle size category, the vertical projection of particles above the bed can be measured as an index of the fish rearing cover provided by flow velocity diversity. This combination of measures would be useful in situations where large particles are flattened rather than round or subround in form. In such cases, relatively unembedded particles (as determined by the measured embeddedness technique) might not create much flow diversity. Vertical projection of cobbles above the bed surface and access to the undersides of cobbles are two important aspects of substrate embeddedness.

The "free winter particle" estimation procedure used by the Clearwater National Forest (Espinosa 1991, 1992) is a method for estimating embeddedness during the overwinter period. This method emphasizes availability of interstitial space in substrate material in the size range 7.6 cm to 1.5 m. A minimum of 10 transects are established at 1-3 m-intervals along a winter habitat area. The percentage of the length of the transects that is occupied by <6.4 mm and 6.4 mm -7.6 cm particles, and free winter particles is recorded. A free winter particle is one that is not totally embedded by fine particles, thereby providing winter hiding cover under the substrate. The percentage comprised by free winter particles is an index to the degree of embeddedness that has significance during winter.

Streambanks and Channel Change

Stream channel change can occur vertically as change in bed elevation (aggradation, degradation), bankfull channel width, bank or channel morphology, and sinuosity. Shifts in bed elevation can be measured via repeated measurements of monumented bed cross-sections. Provided that cross-sections are spaced closely enough, a contour map can be generated that can be compared with a subsequent map to determine total volume of sediment lost or accumulated. This method is useful in estimating net response to a series of discharge events but does not indicate intermediate stages in a dynamic process. Scour chains can provide useful information on maximum depth of scour and subsequent fill but limitations in numbers of such monitoring devices that can be implanted into the streambed make it difficult to accurately generalize about the entire streambed. Lisle and Eads (1991) recommend installing 10 to 40 chains across two to five channel cross sections. The siting of surveyed cross-sections or scour monitors should be made to include several replicated channel unit types that are representative of a reach.

As streambanks are broken and degraded and as sediment is deposited in the channel from bank and general watershed erosion and sedimentation, bankfull width can increase. Unless the channel is highly braided or lined by emergent vegetation, it is usually feasible to read the height of the most recent high flow from the trimline (line demarcated by collection of floated debris and erosion of duff) along the channel margin. Bankfull margin is normally approximated by the trimline and coincides with a rapid break in channel morphology unless the banks have been damaged. Channel bankfull width should be recorded on monumented cross-sections. In cases where riparian vegetation is not excessively dense, bankfull width changes can be determined via aerial photogrammetry (Lawler 1993).

Bank morphology often exhibits a complex series of surfaces and changes in gradient with distance from the stream margin that indicates a history of terrace construction during channel evolution and recent flow regime. Shifts in flow regime (e.g., increases in peak flow frequency or magnitude), removal of stabilizing vegetation in the riparian zone, or livestock damage can alter bank water storage and infiltration capacity, produce overland flow, and can result in bank profiles going from vertical or overhung to shallow dish-shaped. Bank erosion can be accurately measured via erosion pins or terrestrial photogrammetry. Thorne (1981)(as cited by Lawler 1993) recommends use of two or more pins every 1-5 m along a stream reach. Hooke (1979)(as cited by Lawler 1993) recommends reading erosion pins each month and also after each major streamflow event. Erosion pins, however, do not consistently give reliable indications of erosion or aggradation for a great many reasons (Lawler 1993). The difficulty in extrapolating to the entire streambank is great. Terrestrial photogrammetry provides a greatly enhanced ability to measure total volume of eroded or deposited bank. In addition, modern theodolites allow construction of digital terrain models for a bank that can be contrasted with subsequent models to calculate volume of sediment added or eroded (Lawler 1993).

Shifts in sinuosity that occur in floodplains produce shifts in channel type (Rosgen 1985, 1994) that indicate change in water surface profile and probable increased habitat complexity for fish. These changes can be easily detected over periods of years using aerial photo interpretation.

Bedload measurements can now be made with greater ease than previously. Some of the relatively new techniques include use of pressure pillow traps, electromagnetic and acoustic detectors, and automatic weighing systems (as reviewed by Lawler 1993). Some of these techniques may make it feasible to more precisely predict bedload movement as based upon streamflow discharge, velocity, and depth.

Temperature

Monitoring of temperature regimes should be conducted with a network of continuous recording instruments (e.g., Ryan Temp-Mentors or Hobo-Temp by Onset) placed in strategic locations throughout the watershed system—upper, middle, and lower reaches of tributaries and mainstem channels. Monitoring should be conducted on an annual basis. An emphasis should be placed on summer maximum temperatures. However, in addition, from continuous recording, calculation should be made of diel variation, mean daily temperature, number of consecutive days in which temperature exceeds biological thresholds (e.g., avoidance temperatures, growth limits, disease thresholds), and winter low temperatures.

Since temperature monitoring is relatively straightforward and inexpensive, direct network monitoring is recommended as a means to gain both precise data at the project, reach, and total stream system levels. A network of continuous recording instruments can effectively deal with spatial and temporal variability. This will provide useful baseline data and facilitate feedback to riparian and stream channel management. There are some good temperature models; however, they all have shortcomings and require local validation (MacDonald et al. 1991). An effective network of temperature recorders on reach and stream network scales is important for proper validation of temperature models (Bartholow 1989, Mattax and Quigley 1989). Use of video thermography is a relatively new tool that makes it feasible to acquire instantaneous data on water temperature distribution (laterally, longitudinally) in the surface layer for an entire stream system, provided riparian cover does not obscure the stream surface (McIntosh et al. 1995). This technology can provide data to reveal coldwater refugia.

Temperature increases from single activities are measured from upstream to downstream on a particular stream reach. Some of the increase can be considered to be attributable to the activity. However, there are natural processes that can lead to either warming or cooling of water in the reach. If the existing canopy closure is known to be a product of long-term natural processes, the temperature increment per length of reach could be considered to be natural. However, past clearcutting, thinning, grazing, agricultural, or urban use actions that maintain a reduced canopy cover or shift vegetative composition (and thereby canopy cover or bank stability in riparian soils) have likely altered thermal loading properties of the reach. Change from this current baseline can be assessed, but such pre- and post-impact monitoring does not indicate what the pre-management (historic, long-term mean) thermal loading rate was.

Natural processes that can lead to cooling over a reach include groundwater input to the reach. This effect alone can complicate determination of the effect of riparian thinning. For example, after riparian vegetation on a reach has been thinned, there may still be an upstream to downstream temperature decrease due to groundwater entry. The rate of groundwater input (m^3/km) to the reach can be measured as the difference between discharge at the downstream end of the reach minus discharge at the upstream end divided by reach length, assuming no intermediate surface water entry points. Groundwater temperature should also be measured. If the project site had been monitored prior to thinning, water temperature might have cooled in this reach because of groundwater entry or warmed because of solar input or equilibration with air temperatures. Therefore, indication of adverse temperature modification for the reach can be achieved by comparing post-project with pre-project water temperature change over the reach length on days with similar direct solar radiation, air temperature, wind speed above the canopy, and groundwater input rate.

In addition to detecting a management effect from the combination of management and natural effects on a single reach with pre- and post-activity monitoring under similar climatic conditions, the management effect should be examined under a variety of climatic conditions. It becomes more difficult to separate natural and management effects when numerous cumulative impacts occur within a larger stream system. If a temperature standard allows a water temperature increase (e.g., $\leq$0.5°C) from a project area, it is necessary to measure the magnitude of the management effect over the course of the reach. Such a standard should be monitored under extreme environmental conditions to ensure that this standard is not exceeded throughout anticipated climatic variation. This requires continuously monitoring water temperature throughout July and August and also monitoring in multiple-years to include cycles of drought, high air temperatures, and varying cloud cover. The monitoring approach using a single reach at two points in time has the advantage of providing comparable channel morphology, substrate materials, groundwater supply characteristics, and other physical features, allowing the pre-activity condition to be a fairly accurate control for the subsequent post-activity period, except for some characteristics that change with climatic conditions.

Another approach to monitoring effects of riparian projects is to measure a temperature increase over the course of the stream reach within the project area and also in an immediately upstream paired reach (each with no surface water entry points) bordered by a late-successional riparian zone. The advantage is that many climatic variables may be similar for the two reaches, such as air temperature, humidity, wind speed, cloud cover, transmissivity, general soil moisture levels, groundwater storage, and water discharge rate. A disadvantage is that channel morphology of the two reaches may differ, as may substrate materials, initial canopy cover, and groundwater entry to the channel. This provides a point-in-time contrast of reaches in the same stream.

A third monitoring strategy is to compare the paired reaches (project and control) under pre- and post-activity conditions (i.e., at t_1 and t_2). These reaches may or may not be longitudinally contiguous. Consequently, a thorough evaluation of the characteristics of these reaches is needed. The project and control reaches can be initially selected using classification procedures based upon these key physical characteristics. If the control reach exhibits and upstream to downstream temperature increase between t_1 and t_2, the cause could be air temperature differences or other environmental variation. The project reach may be subject to this same background, natural temperature increase but also is subject to management effects. Any environmental differences between the two reaches that do not vary synchronously can lead to misleading analysis. To ensure reliable data analysis, one must accurately determine pre- and post-project riparian cover, groundwater input rate, groundwater temperature, and air temperatures for the two reaches.

A fourth approach would be to select multiple reaches that have reasonably similar physical characteristics but vary in riparian conditions (i.e., span the range from clearcut to dense riparian forest). The upstream to downstream thermal increment that occurs on a single day should be significantly determined by the percentage canopy cover of each reach. Using this method, one would assume that air temperature and cloud cover are similar for all the reaches. Variation among reaches in physical characteristics could undoubtedly lead to unexplained variation in the relationship between canopy cover and thermal increment, but if enough reaches are monitored in this simple manner, management effects may become apparent.

If the condition of an upstream control reach is altered (i.e., the canopy cover has been reduced by management), its water temperature will be elevated. If all upstream cumulative riparian impacts result in a high water temperature through the control reach as an input to a downstream project reach, assessment of the significance of canopy removal in the project reach may be affected. That is, if water temperature is already approaching equilibrium with air temperature because of upstream cumulative temperature increases, another riparian harvest unit may produce a minimal apparent effect in a project stream reach, whereas if the upstream reaches had intact riparian cover and delivered cold water to the project reach, the apparent effect would be greater. This places added importance on selecting comparable project and control reaches and knowing the upstream riparian conditions for each. The general significance of riparian canopy removal can appear through monitoring to be less than it actually is provided that a watershed is already highly developed. In addition, canopy removal can lead to thermal effects that become evident years after the initial canopy removal. That is, the thermal increment that appears after canopy removal can increase in subsequent years if the activity also resulted in bank instability, channel widening, pool infilling by fine sediment, and reduced water depth. The sequence of channel degradation events following riparian harvest emphasizes the need to monitor for temperature effects on an affected reach over several years rather than simply at a point in time to document all cumulative effects.

Holtby (1988) estimated that clearcutting in Carnation Creek watershed caused a 3.25°C increase in water temperatures for August. He was able to distinguish this temperature increase from annual variations by use of regression techniques. He predicted Carnation Creek water temperature from a nearby air temperature station. A multiple regression was used to predict mean weekly water temperature from mean weekly air temperature, month, and logging effects (i.e., proportion of riparian length upstream that had been logged). This allowed the logging effects to be separated and distinguished from the seasonal effects of air temperature in heating the stream. The importance of watershed-wide cumulative temperature effects underscores the importance of monitoring and controlling temperatures throughout the watershed and restoring the entire longitudinal temperature profile. Any attempt to monitor trends in water temperature of salmon-bearing reaches must be accompanied by even greater efforts to monitor trends in non-salmon bearing streams.

There is a tendency in land management to assume that management-caused environmental effects are not significant if it is difficult to distinguish them from natural background. The serious management consequences of relying upon concepts of "range of natural variability" are extensively discussed in CSP (Rhodes et al. 1994). Monitoring programs that are designed upon this premise seldom are afforded the scientific rigor to allow differentiation of causes of temperature increase in a reach. Exceedence of temperature standards (biocriteria that specify optimal conditions for each lifestage) should by itself initiate management response to restore riparian zone condition. In practice, this is rarely done. Managers cite their inability to differentiate natural- from management-induced temperature elevation as justification for assuming that high temperatures may be natural for the reach. Rather than controlling temperature on the basis of an entire stream system, reaches that currently have temperatures below the standard are allowed to have activities that increase temperatures until the standard is reached. This process results in a continual loss in suitable habitat from downstream to upstream caused by increasing temperatures. Trend monitoring could detect long-term headward migration of the critical temperature maximum, but such an effort requires establishing a baseline followed by years of monitoring. A mapping on a stream network basis of all reaches meeting or exceeding temperature criteria for various life stages would be useful in summing the useful habitat length or area for these stages through time in a watershed.

All the aforementioned statistical "concerns" are a product of starting from the assumption that planned management-caused impacts throughout the entire watershed are a given, habitat degrading actions will be allowed because it is assumed that the effect will be small and hard to detect, and the burden of proof involves distinguishing the management effect from the background natural condition. If injurious activities were not allowed, the importance of statistics as a management tool guiding the limits of development would be much less. The management context in the former case is often one of attempting to maximize freedom to allow management effects while keeping instream habitat condition within certain pre-determined bounds. Monitoring under this management context is often oriented toward: attempting to ensure that habitat condition does not exceed those bounds; accounting for the correlation between watershed condition and fish habitat condition; following trends in land use, watershed or riparian condition, and fish habitat condition so that future states can be predicted and possibly avoided; and determining the extent to which habitat degradation reduces fish survival. Conducting effectiveness monitoring on riparian thinning or patchcutting activities likely has an objective such as determining whether the action "significantly" increases water temperatures. Evaluation of effects may need to distinguish statistical from biological significance (MacDonald et al. 1991). Validation monitoring long ago has shown that canopy removal increases water temperature and evaluation of this in an ESA context should lead to the conclusion that this is not an action conducive to habitat restoration.

Monitoring objectives are linked to the overall management context. A management context in which habitat systems are consciously degraded to a certain threshold requires an extensive monitoring program, risk assessment procedures, and economic optimization computations to attempt to manage probabilities of species remaining above extinction thresholds. Rarely are sufficient funds allotted to monitoring to even come close to ensuring that habitat conditions can be balanced so carefully. Land management and monitoring in an ESA context should not follow this conventional process. In terms of water temperature control, land management standards recommended in Rhodes et al. (1994) involve establishment of riparian reserves not subject to even low level risk until instream habitat conditions and water quality standards are met. Under this management scenario intensive application of effectiveness monitoring is not needed except to assess general improvement in water temperature owing to riparian planting or natural recovery. Allowing activities that are known from intensive validation monitoring to exacerbate stream water temperature conditions on a reach or watershed basis is inadvisable if the objective of management is to prevent habitat degradation. If a decision is made to allow an impact of a certain degree, effectiveness monitoring of temperature becomes a management feedback mechanism that assesses whether a completed action succeeded in limiting its effects to the desired degree. For purposes of planning improvement in general stream network temperature profiles, a reach temperature model should be employed to evaluate the likely effects of restorative actions (riparian planting, road relocation and surface revegetation, vegetative channel stabilization) in riparian areas (see Theurer et al. 1985).

Part VI: Fish Community Monitoring

Coordination of Physical and Biotic Monitoring

In order to develop a monitoring plan that is sensitive to the spatial organization of habitat types of various geographic scales and the species and communities that utilize them, it is important to review some of the concepts that have been developed in the literature on fish community- or species-habitat relationships. Sampling units for in-channel physical habitat variables could be selected randomly, on the basis of hydraulic considerations, or other physical factors. However, because fish distribution occurs in relation to distribution and organization of recognizable channel features, a stream monitoring program provides its most useful data when the biotic data collection is structured in a meaningful (e.g., hydraulic or other physically-based) spatial framework relevant to aquatic biota. Trends in species abundance or in the structure of fish communities can be examined at various spatial scales. Appropriate spatial scales for trend monitoring and analysis may be dictated by perceived major breaks in the stream physical system that establish "natural" aquatic units or by spatial units used by a species during all or a portion of its life cycle. Identification of a monitoring unit according to habitat use by the most wide ranging species (e.g., salmon) can then establish the size of the largest habitat unit for evaluating total fish community trends.

Salmon-bearing stream reaches may provide distinctly different environments for the life history stages of salmon species that can be categorized by channel type (Rosgen 1985, 1994) and channel unit type (riffle, pool, glide, see for example, Hawkins et al. 1993). The physical conditions of reaches throughout a stream system and the ability of various life history types of each fish species to utilize various portions of a stream system determines the species biological performances (e.g., growth, survival, reproduction) in geographic locales and also the characteristics of the fish communities.

What makes a species sensitive to disturbance? Many fish, including salmon, are well adapted to living in variable environments. Salmon have high fecundity that allows them to rapidly build population sizes that can fully occupy available habitat and compensate for multiple sources of natural mortality during their life stages. Salmon populations become diminished in size when the magnitude of combined sources of mortality (natural plus man-caused) significantly exceeds the natural mortality range. Loss of carrying capacity, owing to loss of habitat quality/quantity, limits the potential population size that can exist under the best (most benign) mortality rates. This loss in carrying capacity can result from progressive, systemic change in certain variables (e.g., basin-wide sedimentation, global warming, basin-wide reduction in riparian canopy cover resulting in stream water heating), spatially localized changes to critical habitats that represent bottlenecks for certain life history stages (e.g., water withdrawal that might halt upstream passage), or periodic phenomena (e.g., El Niño effects leading to low summer flows and warm air temperature; major channel forming floods) that limit survival by chronic or acute impacts. Chronic impacts to habitat quality/quantity arise by direct action (e.g., canopy removal producing water temperature increases) and also by indirect routes (e.g., as water temperature increases in a reach, predation and competition by warmwater tolerant species on salmonids increases). A high management-caused, background, chronic mortality rate may be the result of generally degraded habitat and be expressed in poor emergence, rearing, and outmigration success on a watershed basis. When these chronic mortality loads to a population are compounded by acute impacts, such as major road failures during major storm events, population size can be greatly affected. Physical alteration of streams during floods is greater when riparian zones have been degraded; also the magnitude of the acute effect is greater in degraded watersheds relative to intact watersheds for the same climatic events.

A Habitat-Based Framework for Fish Community Composition

Salmon habitat in its broadest sense can be taken as the entire watershed. The watershed provides the environment that supports individual species. Because of the overlap in habitat requirements of various fish species, any given position in a stream system typically has the ability to support assemblages of fish species (or communities) and the associated assemblages in lower trophic levels (i.e., macroinvertebrates, algae, etc.) that form the food base. The watershed system produces a hydrologic regime, sediment delivery regime, and water chemistry profile relative to its geology, vegetation, climate, and landforms. These driving variables establish the stream channel morphology and instream distribution of physical structures that comprise channel units supporting fish populations (i.e., the riffle/pool systems). The series of linkages between hydrologic characteristics, watershed structure, stream channel morphology, and fish habitat (see Orsborn 1990) constitute the environmental or habitat template for fish production, survival, and community composition. Anthropogenic alterations in fish habitat at any spatial scale within the watershed from activities can be transferred to instream habitat quality and quantity and ultimately can modify dependent fish communities. The degree of impact to in-channel fish habitat is a function of proximity to the perturbation, magnitude and spatial extent of the disturbance or cumulative disturbances, and also habitat condition and sensitivity. Alterations to the in-channel habitat condition can lead to change in the fish community depending on species' sensitivities to environmental change

The channel unit (i.e., riffles, pools, glides, etc.; see Bisson et al. 1982; Hawkins et al. 1993) is often considered a basic habitat unit. Fish use these differentially depending on life stage, size, and season (Bisson et al. 1982) and many of these units are associated with specific structural elements (e.g., LWD, overhanging banks). Land management effects that alter the qualities of these units or the proportional abundance of units in various reaches will have ramifications to the structure of the fish community and differential population survival for the individual species. Habitat quality is altered by actions that, for example, affect percentage fine sediment in spawning or rearing areas, reduce LWD loading, increase summer water temperature, and reduce pool volumes or depth.

In addition to the riffle, pool, and glide channel units, there are other units that have special significance in the distribution and survival of various fish species and life stages (Schlosser 1991). For example, off-channel ponds or sloughs and mainstem alcoves or backwater units provide rearing areas for species that require slow current velocities juxtaposed with fast velocities and also stable overwintering habitat (Hawkins et al. 1993, Bisson et al. 1982; Cederholm and Scarlett 1982). Off-channel habitats in wet meadows and low gradient floodplains are very productive habitats. Considerable Bonneville Power Administration (BPA) dollars have been spent attempting to restore these elements to stream channels (e.g., Everest et al. 1986; Reeves et al. 1990). These habitats have frequently been filled by heavy sediment loads transported from upstream activity areas, are frequently subject to heavy impact from grazing (Beschta et al. 1991) or rendered unusable due to water temperature impacts.

Microhabitats are the essential lower level units in the habitat spatial hierarchy that determine whether a species, life stage, or size class can inhabit the channel unit. In general, microhabitats needed by various fish species include interstitial space in streambed substrate, overhanging banks, woody debris jams, accumulations of fine organic material, exposed tree roots in streambanks, streambank vegetation (source of hiding cover and refuge during overbank flooding), deep pool zones, and the lee side of boulders. Within a riffle, critical microhabitat subelements of channel units include the substrate composition, embeddedness, percentage fine sediment, presence of overhanging banks, quality of the channel margin, etc. In pools, critical microhabitat subelements include presence of LWD, maximum depth, presence of boulders, embeddedness, overhanging banks, water temperature, distribution of depths, etc. Removal of LWD, streambank collapse or destabilization, filling of interstitial spaces in spawning, spring/summer rearing, or overwintering substrate with fine sediment, elimination of streambank vegetation (standing tree stems and roots, brush cover), and creation of homogeneous water depth distributions all contribute to reduction in habitat diversity, elimination of certain microhabitats, and consequently reduction in biotic diversity (Gorman and Karr 1978, Pearsons et al. 1992).

Land management effects on stream channels and fish habitat vary in type, intensity, geographic extent, and duration and can be cumulative at a reach or watershed scale. Therefore, the current and likely future in-channel habitat conditions cannot be assessed by monitoring simply local in-channel variables or condition of the adjacent riparian zone. Because certain impacts (e.g., nutrient concentrations, sediment delivery, water temperature, LWD recruitment, peakflow or low flow water discharge) arise partially or largely as cumulative effects from the upper watershed to a reach, it is expected that fish communities may be different in reaches sharing many basic characteristics (e.g., similar size and gradient, channel type) if their habitat qualities vary because of upstream land uses.

Monitoring of fish communities involves consideration of the appropriate spatial scale (i.e., from the Snake basin, subbasin, ecoregion, watershed, reach, channel type, channel units, to microhabitats). Various species use habitat at different scales (see Frissell et al. 1986). It is essential to adopt meaningful frameworks for examining communities at any point in time and also through time. Communities have been conceptualized as varying in a downstream direction along a river continuum (Vannote et al. 1980). Alternatively, communities are described with respect to a given habitat unit (e.g., the fish community associated with a riffle/pool complex, a reach, or watershed). Classically, fish communities have been described for "zones" that are primarily classified by channel gradient, stream width, or altitude (Hawkes 1975, 1977). Communities are also described for ecoregions (Hughes et al. 1987) that are essentially biogeoclimatic terrestrial regions that subdivide a river continuum into discrete units.

Whether trends in a community are followed on a portion of a river continuum as a continuous distribution (e.g., headwater streams to 6th order) or by discrete units in which community composition changes abruptly with abrupt physical changes (e.g., gradient breaks), there is considerable debate in fish community ecology concerning the factors responsible for structuring communities at locations along the continuum. The ability of various fish species to form assemblages (groups of interacting species linked in a trophic network), to recover population size rapidly after a perturbation, or to maintain a highly stable density despite environmental perturbations are some characteristics describing the population or assemblage that suggest useful biotic indices for detecting effects of habitat alteration. Community structure indices theoretically reflect habitat quality and habitat structure within the spatial scale monitored.

Given the species pool available for a stream system, the communities that inhabit various habitat units are described as being controlled by either deterministic or stochastic factors. These factors are either physical/chemical or biotic environmental variables (i.e., part of the habitat system at various scales of study) relative to a fish species or community of interest. Deterministic models of community composition frequently use stream temperature as a primary explanatory variable for distribution of native cold-, cool-, and warmwater species. In addition, stream gradient and width have frequently been used as indicators of major discontinuities in fish zones (Huet 1959; Hawkes 1975). Deterministic processes are considered to result in characteristic communities owing to predictable outcomes of competition and predation within a classified habitat template. Deterministic communities are characterized by limited morphological similarity, segregation by habitat, microhabitat, and/or diet, persistence in time, and resilience after disturbance (Grossman 1982, as cited by Moyle and Vondracek (1985)). A tendency to return to a predictable structure after localized defaunation (Meffe and Sheldon 1990) is a strong indication of deterministic processes. The defaunation must not be severe enough to eliminate species from nearby habitats that could provide colonists. Defaunation that is extensive spatially can increase the time needed for community recovery by increasing the distance travelled by colonists.

Deterministic factors controlling fish community composition are considered to be the inherent outcome of biological interactions operating upon a physical habitat template. Communities that appear to be controlled primarily by deterministic factors are framed by habitat elements that can be classified qualitatively, such as a high gradient, coldwater, coarse-substrate riffle community. Assemblages in this sort of channel unit would likely differ from those in a fine substrate riffle. Land management practices that affect major elements of the physical habitat template form a new basis for biological interactions to play out and may provide conditions that select for a different set of species from the stream's species pool. At the scale of the reach or stream, shifts in proportions of channel units, riparian cover, or increases in water temperature provide a new habitat template for fish communities integrated at these spatial scales, affecting anadromous and migratory resident species. These kinds of habitat shifts can favor certain species in the community and select against others.

A set of stochastic factors are also implicated in shaping fish community structure (Schlosser 1985, Grossman et al. 1990). These factors include magnitude of peak and low flows for the current year's species cohort (or for the previous year's cohort), predictability of flow timing in relation to key biological events, seasonal flow variability, and stability (e.g., recession rate; ratio of 7-day low flows with 2- and 20-year recurrence intervals, see Orsborn 1990). Similar indices of the temperature regime might also be linked to observed variability in aquatic community indices. To the extent that a species' abundance or presence in the community any given year is a function of highly variable sequences of events such as flow peaks or coefficient of variation of daily flow (Poff and Allan 1995), land management actions that exacerbate the magnitude of these events or alter the frequency or temporal distribution of these events can be expected to have a significant influence on fish species' survival or abundance and community structure.

Both deterministic and stochastic factors can be considered in classification of habitat. Stochastic factors can be described by statistical indices and ranges of behavior can identify habitat classes. Given an environmental regime (e.g., streamflow) described statistically for a period of years [e.g., Schlosser (1985) described monthly flow variability using coefficient of variation for a 20-year flow record], the dynamics of the fish community occupying a particular habitat (e.g., channel unit, reach, stream) could be considered to be specified. To the extent that the environmental regime is relatively stable when viewed on a particular time scale (assumption of dynamic equilibrium, see Schumm and Lichty 1965, Drury and Nisbett 1971), community performance can be considered to be similarly fixed, but subsuming certain annual variation in community composition (species, relative abundance, etc.).

These deterministic or stochastic controls on fish community composition can be modeled as outputs from watersheds or stream reaches. For example, water temperature can be modeled as a watershed (or stream network) or stream reach variable. A fish assemblage at any particular location (e.g., a reach) can be considered to be an outcome of processes operating at local to watershed to ecoregional scales. The annual temperature regimes characteristic of each reach downstream in a river continuum, owing to the intrinsic characteristics (hydrologic, geologic, topographic, climatic) of the river network and the current spatial distribution of more time variant performances (e.g., hillslope harvest units, sediment sources, riparian condition, etc.), partially determine or makes certain fish assemblages more probable at each reach. For highly mobile fish, it might be more meaningful to consider fish community to be more a stream system property than a reach property. As such, the presence of a species at any reach through time is a product of the spatial distribution of reach types and condition; the condition of each reach type is partially a result of the cumulative effects of upstream impacts at the watershed level.

Salmon Life History and Ecology in Hierarchical Habitat Systems

The Snake River mainstem and tributaries historically were occupied by cold- and coolwater native species, many of which are anadromous salmonids. Resident and anadromous salmonids such as steelhead have generally similar habitat requirements as chinook salmon (Bjornn and Reiser 1991); they are sensitive to elevated levels of fine sediment (USFS 1983; Bjornn and Reiser 1991; Scully and Petrosky 1991; Maret et al. 1993); loss of pools and LWD (Bjornn and Reiser 1991; Reeves et al. 1993); and elevated water temperature (Bjornn and Reiser 1991; Frissell 1992). For this reason, monitoring of resident salmonids has been able to provide additional information on or confirmation of effects of habitat modification on salmon survival without the complications provided by effects of mainstem dams, ocean and in-river harvest, and other off-site sources of mortality. For example, Platts (1974)(as cited by Seyedbagheri et al. 1987) found that resident salmonids had declined in the South Fork of the Salmon River since the catastrophic degradation in 1965, corroborating the role of habitat degradation, in addition to mainstem dams, in decimating the endemic chinook runs. BNF (1993) found that resident salmonids had declined over time in the highly degraded Bear Valley Creek, while they had remained stable in undegraded systems over the same time period, corroborating that conditions in spawning and rearing habitat had a major role in reducing the survival and production of chinook salmon, in addition to the extreme mortality encountered at the hydroelectric system.

The adequacy of protection of endangered spring chinook in the Snake River basin can be verified through biotic monitoring by evaluating population trends of all coldwater fish species inhabiting streams of all sizes throughout salmon-bearing watersheds. Promotion of recovery of cutthroat, sculpin, and bull trout in headwater streams through water temperature regime restoration (among other habitat protection measures) will benefit downstream-rearing salmon populations. Downstream extension of usable habitat for bull trout, for example, will likewise expand usable habitat area for rearing juvenile spring chinook.

Native warmwater species do not occur in the Snake River basin, although many of the typical warmwater species from other regions, such as smallmouth bass, carp, and catfish, have been introduced as exotics to the Snake River basin. Canopy removal and stream channel widening in headwater regions increase solar radiation interception, reduce stream depth, and lead to water temperature warming, thereby increasing headward penetration of exotic warmwater species into former or current habitats of coldwater species such as salmon. Displacement of salmonids (salmon, steelhead, whitefish, bull trout, rainbow trout) and other coldwater species (e.g., sculpins, lamprey) by native coolwater species (e.g., redside shiners, see Reeves et al. 1987) or by exotic warmwater species results in a reduced total usable habitat area for spawning and rearing and, thereby, a diminished production capability. For this reason, trends in habitat quantity and quality can be assessed by monitoring distribution of native and exotic fish species. Community composition of individual reaches is a product of habitat factors and also species interactions, although intensity of interspecific interaction itself is regulated by worsening or improving habitat quality trends.

Salmonid biomass reduction has also been attributed to water temperature elevation. Li et al. (1994) found that elevated stream temperatures in steelhead streams of the John Day River basin resulted in decreased steelhead biomass and also a reduction in food availability as the macroinvertebrate communities of warmed streams provided a greater percentage of taxa that were unavailable to fish as food. Frissell et al. (1992) surveyed fish distribution throughout the Sixes River basin in southwestern coastal Oregon and noted a decline in salmonid abundance as water temperature increased. They detected a temperature threshold of 21°C for coho salmon and cutthroat trout and 23°C for fall chinook, above which fish were rare or absent. In addition, salmonid diversity, as measured by the age classes observed for all species, decreased with increasing temperature.

Use of fish community monitoring as a trend indicator to habitat condition throughout salmon-bearing watersheds of the Snake River tributaries relies heavily on population trends for a very limited species pool. Reimers and Bond (1967) and Wydoski and Whitney (1979)(both as cited by Hughes and Gammon (1987)) reported that the fish fauna of the Columbia River drainage as a whole is similar to that of the Willamette River, one of its major tributaries. In addition to the 28 species they reported for the Willamette River, coho, sockeye, chum, white and green sturgeon, and lamprey can be added for the Columbia. Moyle and Baltz (1985) indicate that most fish communities in western streams typically contain fewer that 10 species and that the composition is fairly predictable. Given the limited species pool, stream characteristics such as gradient, substrate, flow regime, and temperature regime provide unique environmental conditions that select for fish species adapted to these conditions. Li et al. (1987) provided a generalized river continuum for the Columbia River illustrating typical downstream shifts in fish communities with stream size.

Within any stream reach (defined by gradient, mean annual discharge, etc.) shifts in the proportion of habitat units (riffles, pools, glides, alcoves, side channels) and the frequency of these units alters the competitive interactions among species inhabiting the reach and the suitability of the reach for individual species. Fish community composition, then, is partially determined by the available species pool for the region, the general characteristics of the stream system (ecoregion, watershed characteristics), the reach (e.g., channel unit composition, etc.), and by biotic interactions among colonizers.

Because Pacific Northwest streams are relatively limited in species compared with midwestern streams for which the IBI was developed, biologists have pointed out the importance of other sources of biotic diversity in northwestern fish communities that are indicators of environmental conditions (Bisson and Sedell 1984, Frissell et al. 1992, Li et al. 1994). For example, age (or size) class diversity is a sensitive indicator of environmental conditions that a fish population experiences. The age structure of species (or stocks) such as steelhead and spring chinook can be used as an indicator of the relative degree of short-term (i.e., within the life span of the species) environmental stability. These taxa have freshwater stages in the Snake basin lasting from 1 to 3 years. Successful reproduction results in a strong year class of young-of-year fish, given adequate escapement. Year-to-year variation in 0+ age strength is indicative of escapement variation and annual variation in freshwater environmental conditions during incubation and soon after emergence. Annual cycles of flood/drought affect the annual densities of 0+ age fish (Freeman et al. 1988). Irrigation diversion during naturally low flow conditions results in even greater reduction of rearing area, barriers to redistribution and migration, increase in water temperature, increased competition for limited food resources, increased predation, and higher potential for poor growth, death, or disease owing to metabolic stress. The natural amplitude of 0+ age class strength variation is exaggerated by anthropogenic perturbation to flow/temperature cycles. Young-of-the-year fish typically exhibit greater temporal variation than older age classes (House 1995, Peterson and Rabeni 1995).

Evaluation of steelhead age diversity on a reach basis may provide different conclusions than when evaluated on a watershed-wide basis considering the potential shifts in age structure that could occur in reaches of different habitat quality as fish migrate within the watershed. For example, reproducing populations of coldwater anadromous and resident species with multi-year freshwater age classes typically have large numbers of 0+ and decreasing numbers of older age-classes (Moyle and Vondracek 1985, Angermeier and Karr 1986). Low relative densities of 1+ and 2+ individuals during any season are indicative of potential environmental inadequacies. For example, Johnson et al. (1986) found that whereas clearcut stream reaches had the highest densities of steelhead parr (i.e., fish $\geq$1+ age class) in southeastern Alaskan streams during summer, these reaches were not able to support these fish during winter. Fish migrated to old growth stream reaches and to reaches in which riparian buffers were retained to seek deep pools and the abundant cover provided by LWD.

The effect of habitat alteration monitored by shifts in age structure are not apt to be a simple function of the proportion of reaches in various states throughout a stream system. For example, there are important spatial aspects to adequacy of winter habitat in watersheds having cumulative losses of riparian canopy. If canopy loss is extensive within certain tributaries rather than small and dispersed throughout a stream system (or if instream winter habitat elements such as LWD accumulations have been removed from a tributary), it is unlikely that fish rearing in the tributary during summer will be able to effectively seek out remaining suitable old-growth/buffered reaches with abundant LWD for winter rearing. Consequently, survival of parr from one summer to the following spring will likely depend heavily on distances and passage conditions between summer and winter rearing areas as well as on the proportions of summer and winter habitat evaluated for the entire stream system. Survival during this period may also depend upon ability of juveniles to migrate upstream from spawning areas to summer rearing areas as stream water temperatures increase. Because a certain portion of a stream system's fish population will be able to seek winter shelter in reaches that have abundant LWD, monitoring of age structure in these reaches will likely give a distorted perspective on biotic health for the entire stream system due to the tendency of these reaches to attract fish from reaches devoid of LWD. Likewise, summertime monitoring of age structure in reaches having extensive canopy removal could provide a distorted view of this impact on the population because fish are relatively less dependent on LWD during this season. However, reach scale analysis provides information on relative level of support from habitat quality to various life stages by reaches within the stream system. Monitoring of effects of land management on fish populations using age structure data must be conducted at a broad enough scale to incorporate the diversity of habitat conditions (e.g., the spectrum from old growth to clearcut reaches) experienced by a mobile population.

Different age classes of a species frequently occupy different microhabitats or have different diets (Moyle and Vondracek 1985). It is important to maintain diversity of microhabitats to allow age class spatial segregation. Differential anthropogenic perturbation to various microhabitats is a potential source of differential effect on a species' age structure. For example, settling of fine sediments in channel margins during initial stages of increasing stream sedimentation would have negative implications for young-of-year survival (see Moore and Gregory 1988a, 1988b). Increases in cobble embeddedness or filling of pocket pools after a period of sedimentation, reduce holding habitat for adults and summer and winter rearing habitat for juveniles. Anthropogenic effects, on habitat, such as fine sediment accumulation that affect different parts and extents of the life history have the potential to lead to local species extinction, especially when combined with other cumulative effects (Marschall and Crowder 1996).

Many Snake basin streams experience severe winter conditions that are frequently made worse by land management activities. Canopy removal, pool sedimentation, and long-term cumulative loss of LWD operate together to decrease winter rearing capacity. Loss of favorable overwinter physical habitat features (e.g., LWD, unsedimented, coarse bed material, and deep pools), reduced winter temperatures in small streams due to lack of thermal cover by vegetation, and formation of ice reduce the utility of many headwater areas for overwinter rearing. Adverse conditions in these reaches are linked with significant chinook autumn outmigrations (Kiefer and Forster 1991). The fall and winter period is also one in which many anadromous species (e.g., coho, cutthroat, and steelhead) move into non-turbid, stable small tributaries to escape the high flows of the mainstem (Cederholm and Scarlett 1982). Intrabasin migration of anadromous (Cederholm and Scarlett 1982) as well as resident salmonids (Gowan et al. 1994, Young 1994) can cover large distances, making it important to evaluate seasonal habitat quality on a broad spatial scale. The importance of complex, high quality habitat in inducing coho to remain in streams over winter has been well documented (McMahon and Hartman 1989, Tschaplinski and Hartman 1983, Bilby and Bisson 1987).

The fate of salmonid juveniles forced to outmigrate early is not well known. Coho that emigrate to the estuary as presmolts are believed to exhibit very low survival, but rearing in lower river reaches can occur, provided that suitable habitat is available (see review of Bilby and Bisson 1987). Temperature regime has been considered the most profound determinant of life history variation among salmon stocks (Miller and Brannon 1982). Shifts in emergence timing can result in lowered fitness or reduced growth to smolt stage (Miller and Brannon 1982). Johnson and Kucera (1985) observed that subyearling steelhead in tributaries of the Clearwater River, Idaho made a transition from small size substrate in summer to large cobble substrate in autumn. Bjornn (1971) also noted the importance of large substrate size in providing overwinter habitat for chinook and steelhead in the Lemhi River, Idaho. Alterations in the annual thermal regime or fine sediment conditions of substrate can reduce overwinter habitat potential of tributary streams. The availability of replacement overwintering habitat in lower river areas is unknown. Lower river winter rearing potential may not be adequate for these displaced fish because of the widespread deposits of fine sediment in lower river channels (Fulton 1968) or lack of carrying capacity of lower river reaches due to overcrowding, habitat alteration, and reduction in capability to rear macroinvertebrates such as chironomids that serve as food (Coutant 1995). Because of a general reduction in overwinter habitat availability and distribution in many managed tributaries and the unknown fate of juveniles that emigrate in autumn, the spatial problem in monitoring the effect of tributary habitat alteration on population survival is difficult. The numbers and percentage survival of those that overwinter in tributaries and those that emigrate must be assessed. Determination of density and survival of overwintering fish involves watershed-wide analysis.

Sedell (USFS fishery scientist, PNW Station, pers. comm.) observed that the effects of a 100-year flood on increasing channel width and LWD transport in a logged riparian area of the Breitenbush River were significant. However, upstream where the flood passed through a wilderness area, channel width was highly resistant to change. The significance of maintaining stream system stability on a regional basis is very great to ensure that physical states are not altered so drastically that recovery is impeded. This same flood resulted in channel widening and accompanying increases in solar radiation loads (increases in water temperature) throughout the Northwest although this effect was considerably amplified in stream reaches or watersheds having significant management impacts. Such management induced changes also occurred with the flood in the middle Willamette (Lyons and Beschta 1983). Channel changes from the 1964 flood were still in evidence 31 years later when the 1996 flood further perturbed these streams that had not recovered. The increased channel damage occurring in streams previously altered by anthropogenic activities versus that found in undisturbed systems indicates that severity of habitat degradation is heightened by cumulative management effects, thereby increasing recovery times.

Hill and Grossman (1987)(see Freeman et al. 1988) noted that some fish species subjected to high flows were displaced from areas having little refuge downstream to areas with greater refuge. The effects of high flow can vary with species (Gard and Flittner 1974). For example, fish species that feed in the water column are more easily displaced than benthic species (Freeman et al. 1988). Johnson et al. (1986) observed that steelhead were forced to move from degraded summer rearing areas to stable refuge areas in old growth or well buffered reaches to survive winter high flows.

Fish distribution has been correlated with annual variability in timing, duration, and amplitude of water level fluctuations (Freeman et al. 1988). These sources of density-independent control on fish populations vary by species, because of species-specific differences in modes of feeding, egg laying (habitat type, depth of deposition in substrate), size of adults relative to refuge availability (distance to refuges, ability to swim against current, crevice dimensions), and timing and duration of spawning. The effect of water level fluctuations on reproductive success of fish has been reviewed by Freeman et al. (1988). Greater levels of variation in discharge patterns tend to induce greater population variability with short-lived species such as most fish and invertebrates (Grossman et al. 1982, see Freeman et al. 1988; Poff and Allan 1995).

Environmental variability, as measured by coefficient of variation of daily flow, is generally considered to decrease (i.e., the environment becomes more benign) in a downstream direction (Horwitz 1978) on a river continuum and habitat diversity increases (Vannote et al. 1980, Gorman and Karr 1978, Schlosser 1982). The downstream increase in species richness is generally attributed to an increase in habitat diversity and volume (e.g., Schlosser 1982). This longitudinal shift in habitat diversity linked to gradients or abrupt transitions in major continuous environmental factors (e.g., water temperature variation with elevation, tributary entry, or groundwater entry) or shifts in discontinuous environmental factors (rock types) leads to a combination of overall gradual and locally abrupt longitudinal shifts of community composition. Loss of LWD, elimination of side channels, clearing of riparian zones, and diking of floodplains disconnects the terrestrial and aquatic systems, reduces structural heterogeneity of large rivers, and reduces abundance and diversity of juvenile fish (Schlosser 1991). Even without major longitudinal shifts in water temperature for a particular stream reach, community composition shifts from one coldwater assemblage to another in an additive pattern because of increasing habitat diversity and decreasing environmental variability in a downstream direction (e.g., Gard and Flittner 1974). Similar shifts have been reported within warmwater river continua (see Rahel and Hubert 1991).

Biological harshness of water flow and water temperature regimes can be regionalized for a set of streams (Resh et al. 1988, as cited by Detenbeck et al. 1992). Flow statistics can easily be correlated with watershed characteristics (Orsborn 1980, 1990). Jowett (1990) found that watershed characteristics were related to flow variability and hydraulic characteristics in New Zealand streams. These environmental characteristics were clustered regionally and were strongly related to periphyton and trout distributions. In New Zealand Jowett and Duncan (1990) developed a discriminant model with nine environmental variables that provided high accuracy in classifying brown trout and rainbow trout distributions (species, size, and abundance). Fish species distribution was related to climatic (water temperature), geographical, and hydrological factors; fish abundance was determined by factors relating to flow variability, river gradient, in-stream habitat, and the presence of lakes. Although no single model explaining fish biomass or density in terms of habitat condition has been found to be universally applicable (Fausch et al. 1988), there is a great similarity between the New Zealand model and those of many other habitat models developed in the western U.S. (see Fausch et al. 1988). Fausch recommended application of regionalization schemes (e.g., ecoregion, watershed, or stream classification procedures) as a basis for developing more accurate habitat models for predicting fish biomass. The model of Jowett and Duncan (1990) applies a combination of regional level controlling variables and several stream variables to prediction of fish distribution and abundance, thereby employing many of the variables found to be useful in models reviewed by Fausch, but advancing hierarchical systems concepts.

Fish species assemblages have been effectively regionalized based on primary environmental patterns within broad geographic scales. Hughes et al. (1987) was able to differentiate major species assemblages in Oregon streams based on ecoregion classification. The success of such broad-scale models for fish distribution may lie in the ability of regional classification of landform, climate, or soils to differentiate streams among the regions according to associated water flow regimes, water temperature regimes, or other environmental features of fish habitat. Whether models of fish species assemblages or fish abundance are of broad or narrow geographic scope, they tend to have numerous variables in common, such as water temperature, channel gradient, substrate composition, etc. which suggests that many of the variables commonly found to be effective in explaining distribution or abundance should be incorporated explicitly or by correlation with other specified variables in a new, more inclusive model of fish distribution. Reliance only on those factors that happen to explain a large portion of the variability in a limited data set (limited in temporal or spatial scope) insures that the resultant model will be of limited application.

Habitat complexity is a primary factor influencing the diversity of stream fish communities (Gorman and Karr 1978, Reeves et al. 1993). Habitat complexity has been measured along three dimensions by Gorman and Karr (1978). These authors used stream depth, stream bottom substrate, and current velocity as significant elements of fish microhabitat in calculating a habitat diversity index. Schlosser (1982) and Rahel and Hubert (1991) similarly used variation in velocity and depth and substrate types as habitat diversity indices. Schlosser (1982) also included temporal variation of habitat volume at positions along a river continuum as an indicator of habitat diversity. These environmental factors integrate important horizontal and vertical diversity components, at least on a small spatial scale.

On broader scales habitat complexity can be taken as a function of macrohabitat (riffle/pool) frequency and quality and spatial organization of macrohabitat units. Habitat diversity or buffering capacity is created by the effect of channel meandering, large pool occurrence, and distribution of shade (Gorman and Karr 1978). Reeves et al. (1993) found that fish species diversity decreased with decreasing complexity as measured by frequency of large woody debris and pool densities. In addition, they cited the work of several authors who used other indices of habitat complexity such as frequency of habitat units, diversity of substrates, cover complexity, and current velocity diversity. Complexity at an even broader scale (and associated patterns of species additions or replacements) may result from environmental gradients such as those set up by progressive downstream temperature increase or substrate size shifts. All forms of habitat simplification (decrease in habitat complexity) result in decreased fish assemblage diversity.

Habitat complexity indices can be evaluated on a number of spatial scales from channel unit, to reach, to river continuum. Reeves et al. (1993) emphasize that evaluation of habitat simplification at the reach scale may not be sufficient for assessing impacts to fish assemblages. The effects of land management activities can extend well beyond single reaches. In addition, cumulative reductions in habitat complexity on a basin scale (the scale at which many migratory fish species tend to utilize habitat), have integrated effects on fish assemblages and individual species populations at the same scale. Spatial organization of fish habitat on a watershed basis involves assessing features such as distance between spawning areas and holding pools; distance and direction from spawning areas to rearing areas; frequency and spatial distribution of overwintering habitats; frequency and quality of mainstem thermal refuges; frequency of off-channel rearing habitats; distribution of tributary mouths that may serve as summer thermal refuge or sources of drifting macroinvertebrates. Although the influence of habitat complexity at many spatial scales seems to provide logical explanation for fish species distribution, bioenergetics, behavior, life history traits and other characteristics, adequate and sufficient indices have not necessarily been developed. Much clarity in fish population and habitat monitoring programs can probably be achieved by testing the value of new indices of habitat complexity.

Indices of Fish Community Composition

Adaptive land management as proposed in Rhodes et al. (1994) arises from instituting land management standards and monitoring these and various in-channel habitat condition indices. When in-channel CSP habitat standards are not met, or associated, recommended habitat variables indicate negative trends or lack of improvement, either continued application of CSP land management standards or adjustment of these standards is indicated. As an adjunct to watershed or in-channel habitat monitoring, biotic trend monitoring plays an important role in adaptive management. Trend monitoring data on the fish communities associated with available habitat types and habitat quality can provide additional information on progress in reversing habitat degradation. Fish community composition, trends in biodiversity, trends in abundance of rare, threatened, or endangered species, trends in exotic species, and trends in species that are major competitors or predators on T&E species, or indicators of degraded conditions (temperature- or sediment-tolerant species; occurrence of fish diseases; bioaccumulation of toxics in the trophic web) constitute important data needs for providing management corrections because of their collective ability to reflect habitat change at various spatial scales. Fish community composition can be evaluated species by species by trends in abundance and by examining the significance of each species to the fish community, its sensitivity to habitat change (demonstrated by increases in tolerant or decreases in intolerant species), or effect on community stability by native species when challenged by increases in exotic predators or competitors. Community composition is also evaluated using statistical measures and indices for various aggregations of species (e.g., by trophic, reproductive, behavior grouping, native/non-native, etc.). Commonly used indicators include species presence/absence, species richness, species evenness, species diversity, total biomass, proportion of each species (by density or biomass) in the total community, density or biomass (absolute or relative) for each trophic group or reproductive guild, density (absolute or relative) by age or size class for each species, and composition of dominant species. Resh et al. (1988)(as cited by Minshall 1993) lists other structural and functional indicators of environmental impact to aquatic biota, spanning all trophic levels.

Species richness and evenness are combined in measures of species diversity. These measures contain information on numbers of species present and the variability in abundances. Significance of species diversity measures is controversial because the presence or absence of rare species, owing to varying sampling efficiency, can cause marked variation in diversity measures. In addition, presence of common exotic species can enhance species diversity but have adverse ecological consequences. Fausch et al. (1990) and Pimm (1991) recommend against use of species diversity indices because of their lack of biological meaning and statistical flaws.

An index of biotic integrity (IBI) has been proposed as a useful monitoring tool for examining trends in fish community composition (Karr 1981, Karr et al. 1986, Angermeier and Karr 1986) and has gained use in describing macroinvertebrate assemblages (Kerans and Karr 1994). Related procedures are incorporated into standard EPA protocols for sampling macroinvertebrates and fish (Minshall 1993). Weaknesses in the standard RBP procedures (Plafkin et al. 1989), though, make them less desirable as multimetric monitoring tools. For example, they rely upon 100-individual subsamples, which are not apt to be representative of the macroinvertebrate community (Karr 1991); they employ ratio metrics which are more variable and less meaningful than percentages or relative abundances (Karr 1991, Fore et al. 1995); and they emphasize collection from only riffles, which are the most productive habitats and minimize between-site differences (Lenat and Barbour 1994). Because pools may become degraded before riffles, riffle-only sampling causes rapid assessments to be less sensitive to habitat change (Kerans et al. 1992, as cited by Fore et al. 1995).

The structure and dynamics of biotic communities, features accounted for in the IBI, are intimately linked to the overall ecological integrity of stream systems (Karr and Dudley 1981). Rather than focusing NPS monitoring efforts on water chemistry or water quality alone, as in the past, Karr and Dudley recommended consideration of the flow regime, water quality, habitat structure, diversity, availability, and quality of energy sources provided to the stream at various locations along the river continuum as a more complete determinant of ecological integrity and sources of habitat loss, degradation, and fragmentation. The 12 metrics proposed by Karr (1981) as a multimetric rapid bioassessment tool are designed to reflect this complex interaction of habitat qualities. The biological expression of these habitat qualities is observed as species richness, species composition (relative abundance, size frequency distribution), individual health (survival, reproductive status, growth rates), and food-web structure (nutrient dynamics, energy flow) (Karr 1991, 1995).

Hughes and Gammon (1987) modified the IBI for use in the Willamette River and employed the following metrics: number of native, cottid, catostomid, and intolerant species; percentage common carp, omnivores, insectivores, and catchable salmonids; number of individuals; percentage introduced; percentage with anomalies; and total fish biomass (kg/km). Species were assigned to trophic groups and tolerance of pollution (organic pollution, temperature, sediment) from evidence in fisheries literature. Hughes and Gammon adopted 7 of Karr's 12 IBI metrics but substituted cottids for darters; cyprinids for centrarchids; common carp for green sunfish; percentage catchable salmonids for percentage piscivores; and percentage introduced species for hybrids. Cottids, like darters, are riffle-dwellers; cyprinids are good indicators of elevated temperatures; carp are a tolerant species that increase in abundance with degree of pollution. Catchable salmonid presence is a measure that reflects intolerant and piscivorous fish. The presence of introduced species was considered to reflect degree of habitat degradation.

Although the IBI set of metrics for detecting biotic health or trends is a screening method with potential application for detecting excellent, fair, and poor water quality, the IBI is not yet a fully operational procedure for use in the Snake River Basin for either fish or macroinvertebrates because of a current lack of sufficient suitable regional reference collections for streams of various sizes and geomorphic settings (see Fausch et al. 1990). Also, the low number of fish species typical of Pacific Northwest streams limit the application of the IBI in this area (Fore et al. 1995). The IBI integrates numerical, biomass, trophic diversity, species diversity, and pollution tolerance/impact aspects of the fish community, and, consequently, is quite comprehensive in its consideration of ecological effects. Despite its strengths (among them being the ability to integrate cumulative effects), the information loss associated with deriving the composite index can make it inadequate as a stand-alone index (EPA 1993; but see Karr 1991). For example, the IBI has varying degrees of sensitivity depending on type of stressor and this may obscure effects on portions of the community (see Norris and Georges 1993). Two streams may have the same composite scores for different reasons arising from different habitat stresses. The single composite score may be most useful in following trends on a single stream or in comparing paired streams. However, a better understanding of mechanisms of impact can be attained by examination of the component metrics of the IBI, which hold more specific information on response of components of the community (Fore et al. 1995). If care is taken in sample collection and taxa identification and enumeration, even more interpretative power is available by investigation of trends in individual species (R. Wisseman, stream ecologist, pers. comm., 1996).

Li et al. (1987) show a generalized historic distribution of fish species by stream size on a river continuum for the Columbia. Given the lack of high quality habitat and biological communities representative of historic condition, it would not be feasible for the IBI (or the modification used by Hughes and Gammon 1987), based on new collections, to indicate the absence of coho, chinook, chum, and lamprey in many streams that formerly supported them. That is, major ecological changes can be obscured by selection of the reference collections. When salmonid abundance is influenced by stocking, harvest, and off-site impacts, in addition to tributary habitat change, the IBI loses effectiveness (Fore et al. 1995). Also, it is questionable, given the possibilities for species substitutions and the coarseness of the scoring categories, whether the IBI would actually be a useful early warning indicator. However, Steedman (1988)(as cited by Fausch et al. 1990) found the IBI to provide very stable scores from multiple samples collected in a stream and also among years. In addition, scores were highly correlated to indices of watershed development. Although the IBI integrates effects on fish communities, populations, and individual fish, effects to habitat components that lead to impacts on endangered salmon populations may be best monitored biotically as trends in the salmon populations themselves. When salmon are present in very low numbers and the numbers from year to year are highly variable, monitoring of other fish species having similar sensitivity to principal forms of environmental degradation (especially fine sediment and elevated temperature) must be conducted (see Fausch et al. 1990). Components of the IBI may be found that are keenly sensitive to sedimentation and temperature increases. It is probable that trends in sediment sensitive species (or guilds dependent on substrate condition), for example, will not be sufficient information taken alone because sensitivity is often dependent on abundances of certain competing tolerant species (Fausch et al. 1990).

The IBI is a composite measure of fish communities that permits relative comparisons. That is, scores are assigned based on degree of similarity of various components of the fish community to least perturbed reference sites of a certain size stream in a particular biogeographic and geomorphic region (Fausch et al. 1984, Lenat and Barbour 1994). Even though the IBI is a relative measure, its components must be carefully selected to compensate for the fact that many components do not respond linearly over a wide range in environmental conditions (Fausch et al. 1990, Karr 1991). That is, after thermal pollution reaches a certain level, the response of salmonids may no longer provide useful information. Availability of minimally perturbed sites in higher order reaches is typically very limited owing to cumulative effects in the contributing watershed, so reference scores for these historically very significant production areas are apt to be greatly altered. Also, given the linkages between land disturbance and habitat conditions, reaches that have been relatively unperturbed by on-site activities in recent years but which are located in stream systems with significant disturbance upstream would probably not reflect "unperturbed" fish community relationships.

If collections from "least perturbed" but seriously altered large stream reaches serve as the reference for all other more heavily altered reaches of the same type in an ecoregion, the comparisons made may indicate relative degree of change but provide little insight regarding the magnitude of overall change or causes of change. It should not be assumed that the reference condition represents excellent conditions; it is merely a benchmark for one point reached in a long-term degradation trend. This limitation of IBI is shared by all community indices that reflect change from an initial condition. Possible recourse to this dilemma are to reconstruct the reference condition from historical collections; estimate from knowledge of species pools and theory of community assembly what community composition would be in high quality reaches; or explain the distribution and abundance of individual species based on their known ecological requirements and estimate population trends given the expected direction of habitat change under restoration.

Examination of the species list and relative proportions of dominant taxa plus determination of those species that are in exceptionally low abundance or are totally absent might be sufficient to reflect long-term ecological trends. Shifts in the balance of coldwater versus cool and warmwater species or adverse competition from exotic and pollution tolerant species are indicators that can be disaggregated from composite indices such as the IBI. They can provide specific information on trends in temperature or streambed substrate condition in reaches throughout a stream system.

Ecosystem Response to Management Impact: Application of Concepts to Habitat and Community Monitoring and Analysis

Ecosystem health is identified partially in terms of the responses of communities to habitat alteration. Community stability, sustainability, resilience, and resistance are frequently used to describe these responses. Costanza (1992) summarized definitions of key population response terminology from Pimm (1984) and Holling (1986):

> A stable system returns to initial equilibrium after being disturbed. A system might be stable under small perturbation but not large. Strength of the perturbation is a relative term and perturbations to physical habitat are typically complex in effect and heterogeneous in time and space.
>
> A system is sustainable if it can indefinitely maintain its structure and function. Sustainability implies lack of succession and consequently applies to climax systems (Costanza 1992). A more comprehensive view of ecosystems (one including dynamic behavior) would acknowledge that local communities continually shift in composition under successional trends and in response to natural disturbance regimes. A sustainable ecosystem under this concept can be viewed in a larger geographic scope as one allowing for dynamic equilibrium in community composition (e.g., a polyclimax system with successional trends on each site type exhibiting a range of trajectories) when its interacting components are taken as a functional hierarchical unit (e.g., a watershed, a stream system).
>
> Resilience is an index of the rate of return to equilibrium following perturbation.
>
> Resistance is the degree of deviation in structure or behavior of a system after perturbation (Costanza 1992).

In a monitoring context, these concepts of community response to habitat perturbation or of habitat or environmental dynamics are valuable in that they suggest indices for analysis of biotic or environmental data. For example, Poff and Allan (1995) suggested the use of predictability of daily streamflow, baseflow stability, lowflow predictability, or coefficient of variation of daily flows as useful hydrological indices for predicting community composition, guild structure, and species distribution. Horwitz (1978) suggested using coefficient of variation (CV) of daily flows, log of daily flows, proportion of time with no flow, and recurrence intervals of high and low flows as biologically significant indices of hydrologic regime. Pickup et al. (1994) demonstrated that a resilience model of range vegetation recovery after grazing disturbance is an effective tool for monitoring rangeland and severity of grazing impacts on plant cover. With this model the vegetation response in the absence of grazing, as detected by remote satellite sensing, is a function of rainfall magnitude and initial vegetation cover. Communities (fish, total aquatic vertebrates, macroinvertebrates, algae, terrestrial plant cover etc.) can be evaluated in terms of their stability, sustainability, resilience, and resistance to perturbation. The physical habitat occupied by the community should be evaluated according to these indices at the same spatial scale.

Response of the community is linked with response of habitat to perturbation. For example, indices of stability and variation in environmental (habitat) factors are linked to population stability and variation (see review of Grossman et al. 1990). Grossman et al. (1990) recommend use of coefficient of variation calculated over several years of sampling to classify level of population stability. Stability is a relative term that varies depending upon the temporal or spatial scale of analysis and the biotic variable monitored (e.g., population size, community structure, ecosystem property such as community respiration rate). It is also a property of the degree of coupling of a system to other systems at a given level in a spatial hierarchy and the spatial extent and magnitude of disturbances relative to the system size (O'Neill et al. 1986). The greater the degree of coupling between parts of a system on any hierarchical level, the more the behavior of one part affects the other; that is, if one linked part becomes unstable, the other does too. Such system interdependence serves as the rationale for emphasizing channel-riparian linkages in stream management and monitoring. Cumulative watershed sediment delivery has a more diffuse linkage to channel substrate condition so requires more spatially extensive monitoring.

Indices of environmental or related population response that describe resilience, predictability, coefficient of variation, etc. have monitoring value in various ways. As in the use of IBI, it is difficult to know how to evaluate the index except as related to other similar streams. A regression may be developed to relate the index to measures of environmental stress. Trends in the indices for a given site may indicate level of habitat recovery. The indices serve as useful means of classifying stream types so that more efficient statistical stratification of sites can be done to establish monitoring programs. Such environmental indices may also be useful in regressions or multivariate techniques to develop hypotheses about causes for differences in community composition in streams having different environmental regimes.

Individual activities are typically multifaceted in their effects on aquatic habitats (Karr and Schlosser 1978) and heterogeneous in time and space. For example, loss of upstream canopy increases stream temperature, reduces long-term LWD input, increases light input and primary production, shifts the macroinvertebrate food base (community structure and biomass) on a local level, reduces bank stability, and increases sediment delivery. Shifts in hydrologic regime are associated with change in habitat depth, current velocity, food availability, and thermal regime (Poff and Allan 1995). Because of the multiple responses, a return to equilibrium conditions involves many linked changes in habitat condition. A single management action (e.g., riparian logging) can have multiple, linked effects, making the strength of the perturbation difficult to compare with other kinds of management actions that also may have multiple effects. This causes difficulty in devising indices of level of watershed perturbation for correlation with IBI values within an ecoregion. Attempts have been made to convert many types of land disturbance to a common unit of impact (e.g., sediment delivery, see USFS 1981), although other types of impact must be integrated into an index to cumulative impact. Management actions such as stand thinning produce multiple types of habitat alteration and recovery trends that can operate on different time scales. Monitoring of systems having varying levels of sensitivity to perturbation may also provide some notion of sensitivity to restoration measures (resilience), rates, and thresholds of recovery (Wolman and

Gerson 1978), such as how much sediment delivery must be reduced prior to being able to observe improvement in substrate conditions (Scrivener and Brownlee 1989).

Short-term monitoring of habitat alteration and associated biotic trends conducted at the site of a single impact may provide a different picture than monitoring conducted on a more holistic, extensive basis. For example, canopy reduction on short stream reaches has been shown to have certain apparent local benefits, such as increased trout growth and density (Murphy and Hall 1981, Murphy et al. 1981, Murphy et al. 1986). In areas where water temperatures are below the growth optimum of trout and low light levels limit primary production, canopy removal can increase growth, all other environmental factors being equal. Murphy and Hall (1981) found that the initial increase in primary production, macroinvertebrate biomass, and trout biomass attributable to riparian clearcutting and increased solar input is reversed in 10-20 years with canopy regrowth in Oregon Cascade Mountain streams. When canopy cover is reestablished, the negative effects of streambed sedimentation that accompanied logging are no longer masked by the boost in autotrophic production that typically accompanies canopy removal. In the Snake Basin this canopy regrowth likely takes far longer, especially because cleared riparian areas become transitional range for livestock grazing, which further impedes recovery. Trout biomass in reshaded second growth sites in the Oregon Cascades has been observed to be lower than in old-growth sites (Murphy and Hall 1981, Murphy et al. 1981). In addition, as temperature recovers, other habitat factors worsen because the source of LWD recruitment is diminished and its replacement is delayed for decades or sometimes centuries. This effect causes lasting reductions in habitat quality and channel condition. On the basis of water temperatures alone, it is likely that increased water temperatures in headwaters reduce watershed scale fish production because they tend to decrease the total usable habitat area when analyzed on a watershed scale (Rhodes et al. 1994, Everest et al. 1985, Beschta et al. 1987, Theurer et al. 1985). Downstream reaches that had tolerable temperature conditions for fish, may develop intolerable temperatures after the upstream reach is affected. Long-term lack of in-channel LWD in the affected reach also translates effects downstream owing to reduced LWD transport to downstream reaches and reduced total watershed potential for storing sediment. Monitoring programs must be designed to account for local habitat and biotic trends and also distributed and total impact synthesized for an entire stream system. For example, benefits to fish production from actions taken in one reach must be viewed in the context of trends in the total fish production system, and the benefits observed in the short term must be contrasted with those in the long term.

Reduction in habitat complexity caused by anthropogenic perturbation tends to be associated with reductions in fish community diversity (see review of Reeves et al. 1993). The effects of various types of perturbation can vary among species, favoring some species while selecting against others. However, the net effect of habitat simplification is to decrease community diversity. Impacts of anthropogenic perturbation to community structure and composition, assessed at a watershed scale, have been considered by some researchers to be lowly reversible (see review of Reeves et al. 1993). Hicks (1990)(as cited in Reeves et al. 1993) indicates that habitat is generally very slow to recover in complexity to levels found before harvest. Fish species populations affected by habitat quality reduction can experience delayed recovery owing to the slow rate of habitat restoration. For example, Schwartz (1991) (as cited in Reeves et al. 1993) found that cutthroat trout populations in Oregon's Coast Range had not recovered even 25 years after clearcut harvest. Certainly, the time frame for recovery of fish assemblage diversity is at least as great as that for recovery of habitat complexity, the necessary precursory condition.

Detenbeck et al. (1992) found that of all the fish families they considered, centrarchids and minnows were the most resilient to disturbance, salmonids the least resilient. Resilience was lowest (and recovery times longest) in response to "press" (chronic) disturbances, such as mining, timber harvest, and channelization. The mode of reproduction of salmonids may be partially responsible for the more intense negative effect from these types of chronic disturbances owing to the long-term reduction in substrate quality they produce. Fish that require rock substrate for redd construction needed greater time for recolonization and restoration of original population density than species in other reproductive guilds after press disturbances involving substrate modification. The persistence of the disturbance effect or habitat recovery time leads to different effects among species depending on their average life span or age at sexual maturity (Detenbeck et al. 1992). If the disturbance period has a long duration relative to these key life history factors, and is spatially widespread, the effects become chronic. The more widespread a disturbance, the greater the limitation is to any species with low dispersal ability. Under pulse disturbances (limited duration) assemblage composition within any reach may be relatively stable provided that nearby refugia are available or the disturbance is very limited spatially, allowing straying from similar adjacent habitat to occur rapidly after termination of the disturbance.

Recovery times for populations that had been affected by press disturbance vary from 5 to >50 years (Detenbeck et al. 1992). The recovery time is at the low end of this scale when no direct habitat modification is involved. Recovery after pulse disturbances was found to vary from 0.1 to 6 years.

Effects of environmental alterations on single species are of ecological and social concern, just as shifts in community composition are. The ESA is evidence of the management concern for disappearance of species from communities. Species extirpation is of concern for economically important as well as naturally uncommon species. Disappearance of keystone species has been attributed to destabilization of aquatic and terrestrial communities (Mills et al. 1993, Willson and Halupka 1995, Meffe and Carroll 1994). Lack of redundancy in trophic roles within aquatic communities may make species loss a threat to continued functional operation of the community and may impede recovery until the functional role is replaced (but see Schindler 1987). The role of returning adult anadromous fish in stimulating nutrient cycling through trophic levels serving as a food source for progeny of these fish species is commonly cited.

Recovery endpoints can be taken as the time for (1) reappearance of a species, (2) recovery of total fish density, species richness (e.g., at least 80% of predisturbance taxa), or biomass, and (3) recovery of the density, biomass or age structure of individual species (Detenbeck et al. 1992). These biotic indices are of special importance in trend monitoring to assess rate of recovery of species as well as the community. Similar recovery monitoring is appropriate for macroinvertebrate, algal, and riparian plant communities by riparian landtype as affected by varying histories of perturbation. Hughes et al. (1990) consider that macroinvertebrate and algal assemblages may be more useful as subregional reference indices than fish assemblages because of their responsiveness to subregional conditions. In the absence of subregional biotic controls (reference community indices representing pristine conditions), recovery must be tracked by biotic monitoring (fish, macroinvertebrates, algae, riparian plants) emphasizing community and species population response to corresponding habitat condition trends.

Hughes et al. (1990) indicate that fish assemblages are affected by the ecoregion in which a stream is found, proximity to other nearby ecoregions, river basin, species introductions, unusual substrate, springs, migration barriers, confluence with large waterbodies, subregional characteristics, valley type, stream size, and stream gradient. Because of the heterogeneous nature of ecoregions, biotic recovery standards may take considerable monitoring effort to develop. Nonetheless, biotic trends will provide invaluable feedback and interpretative value regarding related trends in habitat quality and may lead to future biotic community standards.

Part VII: Macroinvertebrate Community Monitoring

Introduction

Macroinvertebrate sampling provides a convenient means to assess the biotic response to habitat conditions using a variety of indices. Macroinvertebrate communities respond to changes in many environmental characteristics. For example, macroinvertebrate species distribution and the outcome of interspecific competition can be governed by shifts in temperature regime as a result of bioenergetics (Edington and Hildrew 1973). High water temperature is generally detrimental to Plecoptera, Ephemeroptera, and Trichoptera species. A shift in substrate sediment composition can favor certain species that have adaptations to high levels of fine sediment such as certain mayfly species (Hawkins 1984) and be detrimental to populations of other species that require stable substrate and low embeddedness (e.g., Hawkins et al. 1982). The geographic distribution of water chemistry components, such as hardness, pH, and alkalinity, is important in governing the gross distribution of many macroinvertebrates (Magdych 1984, Pearson and Jones 1984, Townsend and Hildrew 1984, Peterson and Vaneeckhaute 1992). Shifts in nutrient loads such as nitrate and phosphate result in shifts in macroinvertebrate community abundance and diversity by influencing the assemblage types and abundance of periphyton. In a stream continuum unaffected by anthropogenic development, the differences in assemblage composition within any reach are controlled by the channel unit types present. Among reaches arrayed in a series downstream, shifts in assemblages taken from similar channel unit types are controlled by a combination of environmental factors. Among the most important are the parallel shifts in composition of the food base. Hydrologic regionalization can constitute the basis for major geographic distribution of macroinvertebrate species. For example, stream systems having extremely low summer base flows can favor species that either require warm water temperatures and low current velocity and low turbulence or are adapted to avoid high temperatures. Streams with frequent, high magnitude winter flows may favor species that overwinter in streams in the egg stage or inhabit interstices of large, stable cobbles. If such streams, with flashy discharge patterns, maintain channel complexity associated with LWD, deep pools, and unembedded cobbles, they can store and distribute a stable source of detritus, which will support a heterotrophic dominated community. A stream that is naturally flashy but is in a watershed that is heavily logged on the slopes and in the riparian zone may easily lose this channel complexity and those macroinvertebrate species dependent on this food base. Channel hydraulics also affect the distribution of macroinvertebrates within a river continuum and among channel units (Statzner and Higler 1986, Statzner 1988). At the scale of the entire continuum, species distribution is a function of water velocity and turbulence, which are both controlled by the interaction of water surface gradient, water depth, and substrate size and roughness. Given the channel hydraulics, species distribution is additionally governed by food availability and temperature regime.

Macroinvertebrate monitoring has become a common water quality monitoring technique that can be both rapid and cost-effective (Lenat and Barbour 1994). EPA recommends the use of rapid macroinvertebrate sampling in many stream monitoring programs. Rapid assessment techniques are of great utility at Level 1 to provide a synoptic survey of the distribution of macroinvertebrate assemblages down a river continuum. A synoptic survey yields information on the distribution and abundance of species by habitat type (Resh and McElravy 1993). Protocols for rapid assessment involve selecting habitat units, sampling design, collection and enumeration methods, desired level of taxonomic differentiation, desired taxa to enumerate (e.g., will chironomids be enumerated?), biotic index, and statistical methods. These kinds of issues are well addressed in Rosenberg and Resh (1993).

In Level I monitoring the number of habitat types sampled should be limited in order to gain a synoptic analysis for an entire river continuum. Sample riffle habitats or riffles and pool tail-outs. Identify stream width, substrate size composition, stream gradient (field and map measurements) for all sample sites. A recommended sample collection method is to kick sample an approximate, fixed area (e.g., 0.5-1.0 m^2) using a mesh size of approximately 500µm.

Level 3 monitoring is used to distinguish more subtle shifts in biotic indicators so that the effects of fine scale change in land use conditions can be assessed. This will require distinguishing changes of approximately 10% of the mean value. We recommend sampling several diverse channel unit and microhabitat types within various channel types to avoid the pitfall of basing all estimates of impact on biotic change occurring in a single type of habitat. For example, riffle habitats may be most resistant to environmental change caused by sedimentation effects, so riffle-only sampling may not provide a sensitive indicator. Wallace et al. (1996) found that the EPT index for rockface microhabitats was a sensitive indicator of chemical pollution. These microhabitats were also sensitive to drought effects but were insensitive to increased sediment from road building and timber harvest. For certain biotic indicators, annual variations, even in pristine streams, can be very high. This is often attributed to stochastic variation (e.g., associated with flood events in unmanaged streams). Because of the large natural shifts in biotic indicators that may occur annually, it is important to monitor reference streams annually. Managed streams, as well, should be monitored for several years to establish baseline conditions that exist under the likely, more highly variable environmental conditions found in these streams.

Because of the large natural variations that can occur in biotic indicators (e.g., annual shifts in species dominance), the indicators need to be selected to represent biotic change that is ecologically significant and is sensitive to degradation in various environmental parameters. Shifts in channel unit proportions (e.g., riffles versus pools) can only be assessed biotically by sampling all major channel unit types. Annual shifts in macroinvertebrate community indicators in a pristine stream, or among pristine streams of a given type, may indicate stochastic variation where a number of biotic states are ecologically equivalent. That is, any one of a number of species may become numerically dominant depending upon chance events. More stable may be the top ten species classified as common. Also, what is perceived through monitoring as stochastic variation may be deterministic response to undetected annual shifts in fine sediment, temperature regime, and other factors. It should not be assumed that a pristine stream exists under static conditions. To the extent that species are affected by continuous shifts in environmental condition, their populations will reflect a sequence of environmental states and will also be able to represent trends in stream degradation. Variance in pristine systems can be viewed as smaller, however, by summarizing community composition in terms of trends in density or relative proportion of ecological equivalents (species assigned to various guilds, functional feeding groups, indicators of low percentages of fine sediment and cold water temperature, etc.).

To attain the higher level precision in determining differences in biotic response to specific local perturbations or general level of watershed development, other adjustments need to be made in macroinvertebrate sampling. Sampling with a mesh size of approximately 250µm rather than 500µm is needed to more adequately represent Chironomidae and other small invertebrates. Quantitative sampling may be required via use of a Hess sampler or a suction device in deeper water. Taxonomic identification should be made to species level to the extent possible, but minimum levels of identification should be to tribe for Chironominae, subfamily for Chironomidae, family for Diptera (other than Tipulidae and Simulidae), Oligochaeta, and a few Plecoptera and Coleoptera. All other insect groups should be identified at least to genus level (EPA 1993). The number of samples should be adjusted to permit distinguishing a 10% difference at $\alpha = 0.10$ and $\beta = 0.10$. Monitoring should be conducted in two seasons to represent the wet and dry periods (EPA 1993). Monitoring at the end of the wet season in the Pacific Northwest should be able to assess community composition prior to major emergence. That at the end of the summer dry season will assess the product of growth during warm weather after spring egg deposition (McElravy et al. 1989). In each season, monitoring should be conducted in two sampling periods separated by 2-3 weeks, if possible, to adjust for variation in timing in streams being compared. For example, a degraded stream may have a warmer temperature regime than its paired reference streams and may, consequently, have an earlier emergence period. Comparisons among streams should be made for the same season and year within an ecoregion, given the major influence of peak flows and drought conditions in determining annual variations in community indicators such as total number of taxa, mean macroinvertebrate density, mean density of specific taxa, and mean Simpson's diversity index (McElravy et al. 1989).

At Level 3, macroinvertebrate monitoring should emphasize comparative and experimental evaluations (Townsend 1989). Comparative evaluations determine the effects of varying levels of perturbation (Resh and McElravy 1993) and can be revealed in shifts in macroinvertebrate or fish populations or community composition or structure. The stream systems being monitored typically have a varied history of development and natural disturbance that is distributed spatially in different ways among watersheds. Comparisons, then, are made among systems whose historical and recent perturbations were not experimentally planned or synchronous. In comparative studies a large number of sites are compared (Townsend 1989). These sites should be stratified as rigorously as possible, using a hierarchical classification based upon potential, including ecoregion, watershed type, channel type, channel unit, and hydraulic indicators (e.g., Statzner and Muller 1989, as cited by Resh and McElravy 1993). Proper stratification will allow comparison of biotic assemblages or population characteristics from similar environments. Development of a suitable index to degree of perturbation is needed. Often this is devised as an index of urbanization (Steedman 1988), percentage of a watershed harvested, equivalent roaded acres (but see critique of this index in Rhodes et al. 1994). Temporal and spatial aspects of perturbation are often indexed by assessing distance of the perturbation from a live stream or predicted extinction rates of sediment delivery with age of disturbance. However, because portions of aquatic communities can respond in different ways to different types of stressors, there is a danger in using an index based on only one stressor type (e.g., sedimentation) when multiple stressors produce a complex response that varies by watershed or reach.

Monitoring after experimental manipulation is an effective means to investigate potential cause-effect relationships on an individual system or on paired streams. When monitoring a stream subjected to experimental manipulation, it is best to compare before versus after on that stream as well as to compare the experimental streams with a paired control stream(s) so that the effect can be distinguished from annual variation. Comparative studies within a region are effective if the sites are well matched and they were influenced by comparable historical processes (Tonn et al. 1990, as cited by Poff and Allan 1995).

Field experiments should be incorporated into Level 3 biomonitoring. The purposes of this are to determine the response of individual species to single perturbations and identify the physical and biological mechanisms involved in the response. A more complete understanding of these effects would likely lead to better ability to identify species or aggregate taxa that may act as good bioindicators for monitoring.

Types of Collection, Sampling Devices

For Level 1 sampling there are several types of samplers that can provide useful information in rapid analyses. Most rapid assessments suggest evaluating the relative abundances of species in macroinvertebrate assemblages. This objective allows kick nets to be used to sample the substrate within a channel unit. Alternatively, a Surber or Hess sampler can be used to make the collection in shallow water habitats. These samplers have the advantage that they would not tend to miss heavy shelled/cased macroinvertebrates that sink to the bottom but they sample smaller surface areas. In large, deep rivers samples can be collected by Ponar grab (or use of suction dredge devices operating within a chamber enclosing the substrate or by vacuuming the substrate surface and associated crevice habitats (Resh and Jackson 1993)).

Level 1 assessment can also be accomplished by sampling the biomass and composition of drifting organisms. Drift samples should include determination of the total volume of water filtered. The biological rationale for taking drift samples in Level 1 analysis is that the biomass and composition (size, species) of drift determines food availability for juvenile salmon. Drift rates can vary in relation to time of day, changes in water flow, and competition for space or food (Culp et al. 1986, Williams 1990, Angermeier and Carlson 1985, Wilzbach and Cummins 1989, Waters 1969). Prey items that are too small in relation to the size of rearing salmon may incur too great a metabolic cost in capture. Prey, such as snails and certain stone-cased caddisflies, can represent a large biomass, but one that is largely inaccessible as a food base to salmon.

Emergence traps can catch a wide spectrum of the insect assemblages present in the stream. If deployed in various channel units along a river continuum, they can be used to obtain monthly collections of adult insects. Insect emergence timing in relation to salmonid emergence will provide information on the potential species that could constitute the food base for juvenile salmon. Adult insects, in many cases, are easier to identify to lower taxonomic levels.

Specialized habitats can form the basis for sampling specific assemblages along a river continuum. For example, some workers have used cobbles of a specified size range as a sampling unit (Cummins 1994). Measurement of the cobbles allows standardization for surface area exposed to colonization. Riffles containing cobbles of the desired size can generally be found in reaches with similar gradients along a continuum. The suitability of cobbles for colonization can vary depending upon existence of surface pits (as in basalt) or degree of angularity in relation to water depth and flow, a determinant of boundary layer thickness and surface layer turbulence. Large or small woody debris that is relatively stable in the channel can serve as a useful sampling medium to assess the relative abundance of wood gouging or scraping macroinvertebrates. Leaf packs (Cummins et al. 1980, 1989), substrate trays (Rabeni and Minshall 1977, Shaw and Minshall 1980), or algae covered flat tiles can provide artificial feeding substrates for collection of macroinvertebrates. Disadvantages of these artificial substrates are the difficulty involved in placement and retrieval, possible variations in colonization times, and the tendency for these media to become fouled with sediment and otherwise not accurately represent ambient conditions. Except for deep-water habitats, it is probably not efficient to attempt to characterize the primary riffle-dwelling assemblages by focusing on artificial substrates, especially using leaf packs.

Taxonomic Identification

Many rapid assessments feature sample sorting and identification in the field. Identification in the field is normally done at least to the family level, but skilled taxonomists can often identify specimens to the genus level as easily. Species level identification is needed in rapid assessments if community diversity measures are used and are preferred in richness measures (Resh and Jackson 1993). In a review of rapid assessments, Resh and Jackson (1993) found that most sampling protocols recommend that the entire sample be examined, but others suggest that a predetermined number of individuals is adequate to characterize the sample. Hilsenhoff (1977, 1982)(as cited by Resh and Jackson 1993) recommended identification of 100 individuals.

Indices

Common biotic measures used in rapid assessments can be listed in five groups: (1) richness, (2) enumerations, (3) community diversity and similarity indices, (4) biotic indices, and (5) functional feeding group measures (Resh and Jackson 1993). These require (1) counting number of taxa (family, species), (2) counting organisms by taxa and determining relative abundance, (3) combining measures of richness and evenness (individuals/taxon) to derive a composite index, (4) using water quality tolerance values for indicator species to evaluate a reach, and (5) identifying community composition via method of feeding, as indicated by morphological structures or mouthparts and behaviors.

A study of several northern California streams suggested to Resh and Jackson (1993) that the best indicators of environmental impact were richness measures (e.g., number of taxa identified to species, number of EPT taxa, number of families), Margalef's index (a community index), Family Biotic Index, and ratio of scrapers to total number of individuals. They recommend inclusion of measures such as this in a composite index. The Family Biotic Index is calculated as $\sum n_i t_i / N$, where n_i is the number of families, t_i is the tolerance value for the family, and N is the number of organisms in the sample. These authors recommend using combinations of several metrics to characterize a stream but discourage the use of ratios of functional feeding groups. Resh and Jackson (1993) and Lenat and Barbour (1994) suggest that proper classification according to functional group (when used to designate food preferences) is often ambiguous because it can vary by region, stream, or in time. The rapid method of Plafkin et al. (1989)(as cited by Resh and Jackson 1993), which is used by EPA, incorporates eight metrics but this includes a ratio of functional feeding groups.

Part VIII: Statistical Issues in Monitoring

Biotic Monitoring

The objective of synoptic macroinvertebrate monitoring is to assess the distribution and abundances of species and composition of assemblages. In Level 1 the synoptic study acts as a means to establish presence/absence of species or relative abundances. It can paint a broad view to the distribution of biota along a river continuum and detect major downstream trends or abrupt shifts in community composition that may be attributable to shifts in land use or condition. Comparative evaluations (e.g., in Level 2 or 3) are performed by monitoring in an attempt to determine the effects of a perturbation or different levels of perturbation on the biota. This can be accomplished as an upstream-downstream evaluation at one point in time using two adjoining and comparable sites (except for the stressor) or as a before-after comparison on the same site.

The measures chosen generally fall into five primary categories (richness, enumeration, community diversity/similarity, biotic indices, and functional feeding group). It is preferable to devise a measure based on multiple indices to more fully describe the assemblage (Resh and Jackson 1993). The biologic measures must be determined with a preferred level of precision. In order to increase the precision of the estimate of the means, sample site selection is done according to strict criteria to reduce sample variability. Habitat stratification procedures are based upon the assumption that given a common species pool, habitats of a particular class (e.g., riffles, pools) within a hierarchical context will support biotic assemblages that are more similar to one another than they are to assemblages from other habitat classes. Riffle habitats can be expected to vary in their biota along a river continuum. Consequently, other variables should be used to efficiently stratify sample sites, thereby reducing sample variability.

In addition to spatial stratification, timing of sampling efforts can be important (see Plafkin et al. 1989, Cummins 1988, as cited by Resh and Jackson 1993). Sampling periods can be designed to coincide with periods of emergence, growth, maximum size or biomass (Resh and McElravy 1993). Inter-annual variation in water temperatures or flood severity can lead to shifts in time of hatching, emergence, or biomass peaks. If an annual sampling interval is selected for a fixed date, there is a danger that annual climatic variations can lead to increased year-to-year sample variation if no means exists to adjust the sampling date depending upon antecedent conditions. For example, early spring sampling on a single fixed date may not be the best means of estimating population density of young-of-the-year chinook salmon if cold water temperatures persist into mid-spring, delaying completion of emergence.

Replicate samples are required when attempting to precisely estimate mean values for biologic measures. The number of replicates needed depends partially upon study objectives. That is, if the objective is to assess trends at a local site, stratification by channel unit will permit all replicates to be concentrated in a few or a single riffle or a few or a single pool within a study reach. In monitoring of LWD or fish numbers within a basin in relation to land management practices, study objectives may require establishing units of stratification as the entire stream system (e.g., Overton et al. 1993) or the upper, mid, and lower reaches of the stream system (Hankin and Reeves 1988) and having each channel unit of a particular class be a sample. In these cases, replicate sampling is achieved by counting every *k*th unit. Hankin and Reeves (1988) found that fish distribution in pools is highly variable and consequently, it is not accurate estimating total stream fish numbers by extrapolating from counts in a few representative channel units.

Although it is sometimes impossible to devise replicates for certain monitoring situations, the ability to determine valid differences between means depends upon proper experimental design. So, to the extent possible, both spatial and temporal psuedoreplication should be avoided. Spatial pseudoreplication involves comparison of a single reference and test site, even though many samples might be taken at each site. Temporal psuedoreplication involves taking a series of samples through time, each of which is considered a replicate. However, these are frequently correlated and therefore, not independent (Cooper and Barmuta 1993). To avoid pseudoreplication, it is advisable to generate several years of data on several reference streams, where available, taking statistically sufficient replicates in each reference stream. In addition, two or more treatment streams should be examined. In cases where no unmanaged reference streams are available, it may be possible to relate biotic indicators to varying degrees of impact, as inferred from general watershed development, or more directly from stream and riparian condition.

Macroinvertebrate sampling is typically done by disturbing approximately 0.09 to 1 m^2 of stream bottom to a fixed depth. Macroinvertebrates often have clumped distributions that may be caused by behavior, food or substrate (e.g., macrophyte, stable cobble) availability, or other microhabitat-related reasons. Consequently, the size of sample taken in relation to spatial distribution pattern within a channel unit partially determines variability of the biologic measure. Counterbalancing the influence of sample size, the number of replicates for a given sample size increases the precision of the estimate of the mean. Unless a large sample is subsampled (assuming that aggregation in the larger sample is reduced by mixing preserved specimens and that subsamples are randomly drawn) or a fixed number of individuals are counted per sample, larger samples will require greater processing costs and they may not reduce variability as much as by reducing sample size and increasing sample number. The compromise between sample size and number depends upon the total number of specimens obtained per sample and the number per taxon. When mean values become low for a sample, precision tends to decrease. Elliott (1977) suggests that when population dispersion is aggregated a large number of small samples is the most efficient means to improve accuracy. Although this may be a useful rule of thumb, it may be efficient in the long-term to plot variance versus sample quadrat area to determine the scale of clumping and smallest sample size that produces a low variance (Elliott 1977).

A fixed percentage change in mean value can be detected with fewer samples when the mean is large than when it is small (Paller 1995). With endangered species, their low population sizes per habitat unit would result in increasing difficulty in detecting, for example, a 10% change in population size. Sample variance in estimation of population size depends partially upon matching the scale of habitat patchiness with sample size (i.e., area or volume of stream surveyed) (Green 1979, as cited by Paller 1995). Increased precision, lower cost in sampling, and fewer samples required to estimate a mean at a fixed level of precision can be achieved by adjustments in variables measured. That is, in estimating species richness or total fish community biomass, fewer samples are required for a fixed level of precision than if biomass of individual species were estimated (Peterson and Rabeni 1995). Detecting trends in endangered species may require estimating trends in numbers or biomass of individual species, but if objectives permit, sampling effort can be reduced by careful selection of indices. Nonetheless, it must be recognized that different kinds of indices may require different numbers of samples to achieve the same precision.

Estimation of the number of replicates needed per habitat type can be made by applying formulas such as that provided by Sokal and Rohlf (1981)(as cited by Resh and McElravy 1993). This formula is based upon a normal distribution and homogeneity of variances. Other distributions require data transformation (Resh and McElravy 1993) or use of other formulas (Elliott 1977). The number of replicates depends upon the desired difference to be detected and the acceptable risks of making Type I and II errors. A Type I error, signified by the α value (significance), occurs by concluding that two means are different when, in fact, they are not. A Type II error, signified by a specified β level, occurs by concluding that two means are the same when, in fact, they are not. For monitoring of biological indicators (for fish, macroinvertebrates) or habitat condition to be effective as an early warning of adverse environmental change, the higher the precision the better. High precision and small minimum detectable effects generally mean large number of samples and greater costs. The advantage of rapid assessments is that they provide a means to detect change with smaller costs than with quantitative techniques dependent upon estimates of density and identification of species. If the biotic resource is highly valued, it is better to inadvertently produce Type I errors, thereby initiating precautionary land management actions. Reducing the selected α value from 0.05 to 0.20 reduces the significance level of the statistical test of means and increases the probability of finding a difference between reference and management-influenced conditions when there is not. Reducing the β-error increases the power of the test and ability to determine differences between means.

There is no fixed number of samples that can be automatically specified to provide a desired ability to discriminate a particular percentage change in a mean value. Even though the statistical considerations involved in determination of an appropriate number of replicate samples are fairly rigorous (e.g., see Peterson and Rabeni 1995), the number of replicates taken in practice in recent stream monitoring programs, as evaluated by Voshell et al. (1989) (as cited by Resh and McElravy 1993) ranged from 2-30 with most studies taking 2-5 replicates. In estimating community level attributes, such as species richness or total fish biomass, Peterson and Rabeni 1995 recommend eliminating 0+ age class fish from analysis because of their high annual variability. An adjustment such as this can decrease the number of samples required for precise estimates of certain parameters.

A Level 1 monitoring program can provide pilot information that may be useful in assessing the coefficient of variation (CV) for a particular variable, thereby helping to define how many samples need to be taken. Each biological measure used in a multi-metric index can have a different CV. The lower the CV for a particular measure, the lower the number of replicates needed to determine a desired minimum detectable difference between means (Resh and McElravy 1993). For a biomonitoring study of a northern California stream, Resh and McElravy (1993) pointed out that with five replicates per site the minimum detectable difference for four biotic indices (number of taxa, number of individuals, Simpson's Diversity Index, and number of *Baetis tricaudatus*), measured as a percentage difference in the means, reached a value that did not improve appreciably with even ten replicates. Using two years of data, a 56% difference in means was required to confirm a difference at a 5% level of significance for number of taxa and using a median CV value in the equation of Sokal and Rohlf (1981)(see Resh and McElravy 1993). Bisson, Gregory, and Nickelson (1994) reported that interannual CV for juvenile coho, steelhead and sea-run cutthroat was 50% and that for resident cutthroat was 25% in stream studies of the Northwest. These authors used this information to estimate that detection of significant changes in population abundance would require from 20 to 600 years of pre- and post-treatment monitoring. With a significance level of 0.05 and a power of 80%, a CV of 50% would require a sample size of approximately 15 to detect a change of 50%; a CV of 25% would require a sample size of approximately 10 per group to detect a change of 25% (McDonald et al. 1991). A change of 15% could be detected with these statistical parameters if CV were decreased to approximately 15%. If no data are available to assess CV, it is generally reasonable to take a sample size of approximately 10 (L. Conquest, pers. comm., as cited by McDonald et al. 1991).

In rapid assessments an initial simple comparison made between the reference and managed stream is made by calculation of percent similarity. This is calculated as the (managed/reference condition) x 100. Statistical tests of significance of difference between the biological indicator for reference and managed streams generally include use of analysis of variance or t-tests. It is typical to log-transform enumeration data and to arc-sine transform percentage data (Resh and Jackson 1993).

Analysis at Level 3 may take the form of performing t-tests or an ANOVA to detect differences in biotic indicators among reference and managed streams. Assumptions with these parametric analyses are that error terms are normally distributed, treatments are independent, and variances homogeneous (Cooper and Barmuta 1993). If level of perturbation can be quantified and there are more than about 5 treatments with replication at each treatment level, regression techniques can often express the treatment effect better than ANOVA, which may require more replication (Graney et al. 1989, as cited by Cooper and Barmuta 1993). Correlational data, however, can only suggest causal mechanisms. In addition, biotic response of two streams may be the same, but for different and unknown reasons, making it difficult to interpret meaningfully (Cooper and Barmuta 1993). Temporal and spatial trends in macroinvertebrates can often be understood better by use of multivariate procedures rather than univariate methods like ANOVA which depend upon reduction of many variables into summary indicators, such as total number of taxa or individuals. Compensating trends in multiple variables can easily be obscured in the process of aggregating data into indicator variables (EPA 1993). Multivariate procedures allow the presence/absence or abundance of each species to be a separate variable for a site or point in time so that trends can be revealed (Norris and Georges 1993). Numerous multivariate methods, such as cluster analysis, ordination, principal components analysis, discriminant analysis, and detrended correspondence analysis are frequently used to give heuristic understanding of similarities among communities in reference and degraded streams or to depict the relationship between community composition and environmental gradients (Fausch et al. 1990). Multivariate statistics are best used to develop hypotheses about the relationships between biotic and environmental trends or relative degree of similarity among groups.

Physical Habitat Monitoring

Selection of multiple sites for monitoring is one of the initial statistical considerations given to applying any monitoring method. Samples should be stratified according to classification principles such as those described in this document so that meaningful comparisons can be made for sampling units experiencing a different levels of perturbation or among replicates experiencing the same degree of perturbation. Any method that employs strict biological or physical criteria in site selection (i.e., sampling substrate percentage fine sediment only within redds or egg pockets, historic spawning areas, or only riffles having certain depth and water velocity conditions) exerts a strong bias to the data. It must be assessed whether by imposing strict conditions on sampling sites only the highest quality habitats are observed amongst a generally low quality stream channel condition. For example, if intended monitoring sites are those selected by salmon for spawning, it is likely that salmon select areas with the best available substrate conditions. Instead of explaining fish-use site selection criteria, one could sample riffles or glides, channel units that typically provide depth and velocity combinations that create suitable spawning habitat. As a channel undergoes degradation by sediment deposition, there may continue to exist spawning areas of suitable condition (although fewer in number). Change in quantity, as well as quality, of habitat of various types is of major biological significance. Quantity <u>and</u> quality can be assessed better by transect methods using regularly spaced sample sites. It is also important to sample fixed, historic spawning areas over the long term so that change in quality will simultaneously reflect change in quantity. An alternative is to monitor a fixed reach of a certain channel unit that historically provided key spawning area, measure trends for area of spawning habitat, and then monitor sediment quality on transects within areas designated as spawning habitat.

The number of samples required to detect a desired percentage change in habitat variables (just like biotic variables) depends upon the statistical variability in each habitat variable. Overton et al. (1993) examined differences between means of habitat variables monitored in Boulder Creek (intensively managed) and Rapid River (primarily undeveloped), Idaho. They classified samples by Rosgen channel type and then by habitat type, using a modification of Bisson et al. (1982). Their data on maximum habitat depth, percentage fine substrate, pocket pool frequency, pocket pool depth, pool volume, and single LWD frequency (no./100m) exhibited CVs of 31, 64, 67, 28, 91, and 140%, respectively for Rapid River by sampling all habitats in the study area. The managed stream had greater CVs for every variable than found in Rapid River. These authors recommend sampling approximately 75-117 habitat units on B-channels (Rosgen designation) to detect habitat differences between streams for all variables that they measured except for pool volume and LWD frequency. To detect a 10% change in mean percentage fine sediment ($\alpha = 0.10$, $\beta = 0.10$) would require approximately 550 samples (each sample is a visual analysis of substrate composition for an entire habitat unit). A 20% change would be detectable by sampling approximately 150 habitat units. These estimates were made by Overton et al. (1993) using a formula of Parkinson et al. (1988)(as cited by Overton et al. 1993) that depends upon CV, desired percentage change detectable, and α and β levels. For pool volume and single LWD pieces/100m they determined that it was necessary to take a 100% sample of the 12 km study reach. LWD, in particular, was distributed in a very patchy manner, leading to high variability by habitat unit.

Bevenger and King (1995) used a stream classification to stratify sampling of stream substrate using the Wolman pebble count procedure. Their classification was based upon channel gradient, watershed area, landform, lithology, and channel pattern. Using a statistical formula of Fleiss (1981)(as cited by Bevenger and King 1995) they determined that to detect a 10% increase in fine sediment (<8mm) with risk levels of $\alpha = 0.05$ and $\beta = 0.20$, it was necessary to have n_r (sample size of reference condition) = 300, n_s (sample size of study condition) = 115. If $n_r = 150$, $n_s = 215$. This indicates that by applying more sample effort to the reference condition, fewer samples are needed on the study site to assess statistical differences. For best sampling efficiency in comparing reference sites to managed sites (assuming they are properly classified and matched to reference sites), a greater effort should be applied in sampling the reference condition so that significant differences can be determined. However, as shown by Overton et al. (1993), the CV for managed sites may be much greater than the reference site.

The number of samples needed to establish the mean ± 5% can be determined using a formula provided by Skille and King (1988), assuming that measured substrate values are normally distributed. The formula is $n = t^2 s^2 / E^2$, where n = number of samples, t = Student's t, s = standard deviation, and E = level of precision. This formula is one provided by Snedecor and Cochran (1967) (as cited by Simonson et al. 1994) for which the normal deviate corresponds to α (significance level) and β (the complement of the power). Sample number can be evaluated for the reach (designated by channel type) as a whole or for each channel unit type. [Note: This implicitly assumes a normal distribution of substrate values, which is often not valid. Most particle size metric values are skewed; hence, the use of log-normal or other skewed distributions.] Using the method of Skille and King (1988), one can consider the reach, identified by channel type, as a unit and sample 3 hoops at each of 10 transects. For randomly selected sample sites on the transect having depths >0.45 m or with >50% of the hoop occupied by exposed rock, Skille and King recommend randomly selecting new sites meeting depth and exposed rock criteria. Reaches for which embeddedness was measured were also required to have an effective gradient of <3%. If the channel type is very heterogeneous and greater accuracy could be achieved by stratifying the channel into channel units, use statistical considerations of the Hankin and Reeves sampling approach for selecting channel units if the channel unit distribution is complex. Take 3 random samples per channel unit and at least 20 samples total per channel unit type in the reach.

Simonson et al. (1994) used the formula of Snedecor and Cochran (1967) to calculate the sample numbers required to detect a percentage difference for numerous habitat variables in 58 Wisconsin streams. They sampled streams of first through fifth order. Reach lengths were 35 or 40 mean stream widths (MSW) in length. The "true"estimate of the mean for each habitat variable (e.g., percentage bank erosion, percentage sand, percentage shading) was based upon 40 transects spaced every 1 MSW. This was compared with estimates based upon 20 transects spaced at 2 MSW, 13 transects at 3 MSW, 10 transects at 4 MSW, and 8 transects at 5 MSW. All habitat variables were described by a single value for each transect. The authors found that the magnitude of change detected with 20 transects was little different from that detected using 40 transects but the change detected with fewer than 20 samples dropped off considerably.

Cattaneo and Prairie (1994) recommend use of mean-variance relationships in sampling water chemistry to estimate the number of samples needed to achieve a desired level of precision.

They described the mean-variance relationship with a power function of the form var = αX^{β} for various chemical constituents. They found that nutrients, such as phosphorus, were the most temporally variable. Stream water chemistry varies greatly depending upon water discharge. However, in the effort to describe water chemistry during the base flow period when much biological sampling is done, Cattaneo and Prairie (1994) reported that 1 to 12 samples are required to estimate the mean with $\leq$20% error in Quebec streams. Given the low variation in summer discharge in streams of the Pacific Northwest, this same relationship may hold for the Columbia River basin. Water chemistry also can be expected to vary spatially within a stream system according to river continuum theory (Vannote et al. 1980). Water chemistry dynamics during storm events or over the course of the annual hydrograph need to account for variation in discharge.

Part IX: Epilogue

In today's climate of threatened and endangered species, appeals, litigation, and controversy associated with management and stewardship of public lands, there is a profusion of rhetoric and "happy talk" emanating from the agencies that purport to change, "reborn" holistic awareness, and sensitivity. Ecosystem perspectives, assessments, and management are being marketed by the land management agencies**.** This "***new***" stewardship philosophy is being touted by the agencies as a dawn of increased awareness, sensitivity, and consideration. PACFISH (USFS and USBLM 1995), FEMAT (USFS et al. 1993, USFS and USBLM 1994) and subsequent ecosystem assessment efforts such as the Interior Columbia Basin Ecosystem Management Plan (ICBEMP) are recent USDA Forest Service examples of the "***born again***" land management philosophy. PACFISH, an interim strategy and template for future planning efforts, exemplifies problems with USFS current efforts to protect aquatic resources. Although it definitely reduces the latitude for degradation, PACFISH is a weak and ambivalent attempt to protect anadromous fish and is unlikely to be effective (see Rhodes 1995). The substance of PACFISH is that it is a cosmetic clone of the ***"best management practices" (BMPs)*** scenario. The BMP strategy of management has been an abysmal failure in the protection of anadromous watersheds and fish habitats. The effectiveness of BMPs (and especially that of combined actions) has not been shown to lead to achievement of water quality goals, nor do BMPs place ecological integrity as an objective (Karr and Dudley 1981). Restoration of stream systems and the endangered species that depend upon them, will require management to improve total aquatic biointegrity (Karr and Dudley 1981). This will require adoption of in-channel and land management standards, such as recommended by Rhodes et al. (1994) that address key elements of ecological integrity: water quality; flow regime; habitat structure; energy sources; and biotic interactions. This involves control of sediment and nutrients; peak and low flow extremes, maintenance of natural flow regimes; maintenance of pools and riffles, sinuosity, cover, and substrate composition; and intact riparian areas providing natural loading of LWD, CPOM, and solar radiation typical of the position in the river continuum (Karr and Dudley 1981, Karr and Schlosser 1978). Flow requirements must consider more than minimum instream flows. They must provide flows for channel, floodplain, and riparian maintenance (Gebhardt et al. 1989, Hill et al. 1991). Restoration of cottonwood forest ecosystems can only be accomplished by ensuring sufficient summer flows, gradual flow reductions after spring peaks rather than rapid drawdowns associated with irrigation or dam operation, and dynamic flow patterns similar to natural regimes (Rood and Mahoney 1992). "Best management systems" must be developed to holistically reach the goal of ecological integrity, rather than applying untested BMPs in a piecemeal fashion (Karr and Dudley 1981). No amount of monitoring can be performed that will protect biotic integrity or provide rapid feedback to avert biologic impact when there is a decision at the outset not to apply our best scientific understanding of stream system health and linkages between the watershed, riparian zone, and stream channel.

In PACFISH, guidelines and management objectives have taken the place of management standards; there is no longer a stated intention to constrain management by monitoring results indicating non-compliance with standards. Given the complexity of natural systems and their responses under management activity, and variation in capacity for recovery, it is recommended that land management should adopt biologically-based habitat standards, conservative land management standards, a rigorous monitoring program, and a framework for making necessary management adjustments based upon data analysis and habitat standards. In PACFISH, r iparian management objectives (RMOs) do not include sediment as a critical parameter and can be characterized as ***"late warning"*** in terms of protection and feedback. PACFISH requires only implementation monitoring; effectiveness and validation monitoring are not required (Rhodes 1995). In FEMAT, fine sediment is noted as a key habitat quality variable but no commitment is actually made to monitor it (Rhodes 1995). The ***"bottom line"*** to PACFISH and FEMAT is the multitude of loopholes in the management direction that decision-makers can use to avoid accountability and conduct ***"business-as-usual"*** activities. For example, in FEMAT there is no need for any project action to be consistent with direction provided by a watershed analysis (Rhodes 1995). Fine sediment standards have not been set forth in PACFISH or FEMAT despite the fact that fine sediment deposition in spawning and rearing habitat is one of the most pervasive sources of degradation from land management. No recommendations have come from the USFS (see USFS 1994) to monitor fine sediment in watersheds of Idaho supporting listed salmon (see Appendix E). PACFISH is superficial, illusory, and displays a lack of firm commitment to protect fish resources. More important, it will serve as a weak template for future planning endeavors that will replace it.

A companion effort to PACFISH is the Section 7 Monitoring Protocol for the Upper Columbia River Basin (USFS 1994) that is being extolled as meeting ESA Section 7 monitoring requirements. Again, under peer analysis—the effort falls well below a standard of solid science and sound logic. The interim salmonid parameters chosen for monitoring are best described as ***"late warning"*** in concept. By the time one can detect a shift in pool frequency or width-depth ratio, the watershed and fish habitat have experienced major degradation. In a watershed where a listed stock is barely hanging-on, this kind of consideration could mean extinction. Sediment, a key and critical habitat parameter, has been excluded from the list. How this will protect watersheds in the Idaho batholith of the Snake River subbasin is beyond comprehension.

Another significant transgression in the USFS (1994) methodology is that the monitoring is only to be conducted in unconstrained C-type channels. A significant amount of the salmon habitat in Idaho and elsewhere is also found in constrained B-type channels. Impacts associated with development occur in these salmon watersheds. Under the USFS protocols, monitoring will not take place in these drainages.

Another feature of the USFS monitoring scheme is the ***"natural range of variability" (RNVA)*** spin that would be used in data analysis. The weaknesses in this approach to monitoring have been extensively criticized in the Coarse Screening Process (Rhodes et al. 1994). It is important to note the potential harm to listed stocks of salmon that can be caused by this foundation for monitoring and management. This concept has been misidentified by the Forest Service. It should be described as the ***"range of existing variability"*** approach. The Forest Service has no idea what the ***"natural range"*** of any variable is! Short-term ***shotgun*** sampling and ***SWAG*** interpretation certainly do not qualify as ***Science***. Few, if any, watersheds in the Snake River subbasin, can be called ***"natural."*** Watersheds in wilderness and roadless areas have been subjected to unnatural fire management for decades. The so-called ***"forest health crisis"*** is a product of that type of fire management. The application of the RNVA approach, as prescribed in USFS (1994), as a monitoring tool will provide an unlimited amount of ***"hiding cover"*** for those land managers that want to escape accountability for their actions. This approach will provide a ***"free banquet"*** for the ***"developers"*** and most certainly will not protect endangered stocks of salmon.

There is a wealth of serious problems associated with recent management products touted by the land management agencies as adequate, sensitive efforts to improve treatment of watersheds and fish resources. With a modicum of scrutiny, none of them holdup to the requisite ESA standard of ***"no additional risk to the fish."***

On February 1, 1994, Dr. R.W. Gorte, a Specialist in Natural Resources Policy with the Congressional Research Service (CRS), U.S. Library of Congress, testified before Congress on "***reinventing***" the U.S. Forest Service. His testimony (Gorte 1994) was based on a 14-month study investigating Forest Service planning, activities, and related problems. The study focused on the National Forest System including many of the concepts, inventories, monitoring programs, targets, and incentives of management. Dr. Gorte's study also identified considerations that would change Forest Service direction, management, and oversight.

One of the major findings of the study that has particular relevance to the plight of the Snake River Salmon, the Endangered Species Act, and the Coarse Screening Process is that ***"monitoring of the implementation of plans and decisions is inadequate."*** Other findings that influence the monitoring situation were:

- the Forest Service emphasizes timber and other physical outputs in planning and decisionmaking
- budget decisions can overwhelm planning decisions and national direction can nullify local decisions
- forest planning gives relatively little attention to ecosystem conditions.

Gorte's study looked at the Forest Service's efforts at monitoring in some detail. He characterized the monitoring efforts in the following way:

- monitoring of National Forest planning and management has been inadequate
- baseline information on preexisting conditions is lacking
- the Forest Service is unable to monitor changes in ecosystem conditions because it lacks baseline data
- there is a lack of incentives to monitor or conversely, there is a lack of penalties for not monitoring. Forest Service managers are evaluated primarily based on achieving "hard" timber targets, spending money as appropriated and allocated, and preventing problems from reaching the "boss"
- monitoring that shows degraded conditions or unbalanced management emphasis that reflects badly on the Agency and its managers is not well-received
- monitoring gets in the way of achieving hard targets and raises attention to problems
- Forest Service managers have a distinct disincentive to monitor the implementation of certain forest activities.

Dr. Gorte summarized his study and testimony by stating that ***"monitoring and public reporting of the results of Forest Service activities are necessary to determine whether the goals were achieved, and if***

not, whether the deficiency occurred because the plan was not, or could not be carried out." As with management direction, monitoring can be most effective if it looks at all the values associated with forests and rangelands. Dr. Gorte emphasized the importance of public reporting of results that contribute to effective performance. This "feedback loop" can facilitate accountability for managers, and it allows comparisons to identify effective performance. However, too often the process has been ignored or rejected. The essence of Gorte's study and testimony (1994) is that significant reform of land management agencies is required to improve Agency performance. A large part of that proposed reform deals with the ***"reward system"*** that over-emphasizes commodity outputs at the expense of ecosystem conditions and other values of public lands. If watersheds and habitats of Snake River Salmon are to be protected and allowed to recover, the ***"reward system"*** must be changed and accountability has to be made a significant part of the monitoring and evaluation process. Without change and commitment, there is little hope that quality management of ***all*** resources will take place on a consistent basis.

Assistant Secretary of Agriculture, Mr. Jim Lyons, on a recent review trip through the Northwest, summarized the monitoring situation as follows: ***"We've always done a horrible job in monitoring the impacts of management activities. We need to have a better handle on what the resource condition is, the status of watersheds and past management practices (Spokesman-Review, Spokane, Washington, June 4, 1994)."***

The Eastside Forests Scientific Society Panel (Henjum et al. 1994) concluded that a comprehensive biological monitoring program needs to be established for national forests east of the Cascade Crest in Oregon and Washington. They stated that "Data on a broad range of biological conditions within eastside forests are simply not available. This shortfall, added to **inconsistency** in what data are available and **inadequate synthesis** of those data, prevents comprehensive assessment of resource condition and poses a challenge to resource managers. Moreover, such inadequate monitoring of public lands suggests a cavalier attitude toward public resources on the part of the federal government that poses a **barrier to public trust**."

As a concluding statement, we might add **something significant needs to be done about it!**

Literature Cited

Angermeier, P.L. and J.R. Karr. 1986. Applying an index of biotic integrity based on stream-fish communities: considerations sampling and interpretation. N. Am. J. Fish. Management 6:418-429.

Angermeier, P.L. and J.R. Karr. 1994. Biological integrity versus biological diversity as policy directives. BioScience 44(10):690-697.

Angermeier, P.L. and P.C. Carlson. 1985. Effects of season and substrate on availability of drift for fish in a small warmwater stream. Trans. Illinois Academy of Science 78(3-4):199-206.

Austen, D.J., D.L. Scarnecchia, and E.P. Bergersen. 1994. Usefulness of structural and condition indices in management of high-mountain stream salmonid populations. N. Am. J. Fish. Management 14:681-691.

Bailey, R.G. 1976. Ecoregions of the United States (map). USDA Forest Service, Intermountain Region, Ogden, Utah.

Bailey, R.G. 1980. Description of the Ecoregions of the United States. USDA Forest Service. Miscellaneous Publ. No. 1391. Ogden, Utah.

Bartholow, J.M. 1989. Stream temperature investigations: field and analytic methods. Instream flow information paper no. 13. Biological Report 89(17). U.S. Fish and Wildlife Service. Fort Collins, Co. 4512 McMurray Avenue, Fort Collins, CO 80525-3400

Bauer, S.B. and T.A. Burton. 1993. Monitoring protocols to evaluate water quality effects of grazing management on western rangeland streams. Idaho Water Resources Research Institute, University of Idaho, Moscow, Idaho. Submitted to U.S. Environmental Protection Agency, Washington, D.C. USEPA Region 10, Seattle, WA. EPA 910/R-93-017. 179 p. plus appendices.

Beschta, R.L. and J. Weatherred. 1984. TEMP-84. A computer model for predicting stream temperatures resulting from the management of streamside vegetation. WSDG-AD-00009. USDA. Watershed Systems Development Group. Ft. Collins, Colorado. July 1984.

Beschta, R.L. and W.S. Platts. 1986. Morphological features of small streams: significance and function. Water Resources Bulletin, Vol. 22(3):369-379.

Beschta, R.L., R.E. Bilby, G.W. Brown, L.B. Holtby, and T.D. Hofstra. 1987. Stream temperature and aquatic habitat: fisheries and forestry interactions. p. 191-231. In: E.O. Salo and T.W. Cundy, editors. Streamside management: forestry and fishery interactions. College of Forest Resources, University of Washington, Seattle. Contribution No. 57. Proceedings of a Symposium held at University of Washington, February 12-14, 1986.

Beschta, R.L., W.S. Platts, and B. Kauffman. 1991. Field review of fish habitat improvement projects in the Grande Ronde and John Day River Basins of eastern Oregon. BPA Project No. 91-069, Bonneville Power Admin., Division of Fish and Wildlife, Portland, Oregon.

Bevenger, G.S., R.M. King. 1995. A pebble count procedure for assessing watershed cumulative effects. Research Paper RM-RP-319. USDA Forest Service, Rocky Mountain Forest and Range Experiment Station, Fort Collins, Colorado. 17 p.

Bilby, R.E., and P.A. Bisson. 1987. Emigration and production of hatchery coho salmon (*Oncorhynchus kisutch*) stocked in streams draining an old-growth and a clear-cut watershed. Canadian Journal of Fisheries and Aquatic Sciences 44:1397-1407.

Bisson, P.A. and J.R. Sedell. 1984. Salmonid populations in streams in clear-cut vs. old-growth forests of western Washington. p. 121-129. In: W.R. Meehan, T.R. Merrell, Jr. and T.A. Hanley, (eds.). Fish and wildlife relationships in old-growth forests. Proceedings of a symposium. April 12-15, 1982. Juneau, AK. American Institute of Fishery Research Biologist.

Bisson, P.A., S.V. Gregory, and T.E. Nickelson. 1994. Interannual variation in salmonid populations: implications for long-term monitoring. Paper presented at North Pacific International Chapter, American Fisheries Society, February 9-11, 1994. Wenatchee, WA.

Bisson, P.A., J.L. Nielsen, R.A. Palmason, and L.E. Grove. 1982. A system of naming habitat types in small streams, with examples of habitat utilization by salmonids during low streamflow. p. 62-73. In Armantrout, N.B. (ed.). Acquisition and utilization of aquatic habitat inventory information. Proceedings of a symposium held 28-30, October, 1981, Hilton Hotel, Portland, Oregon. Western Division, American Fisheries Society. Bethesda, Maryland.

Bjornn, T.C. 1971. Trout and salmon movements in two Idaho streams as related to temperature, food, streamflow, cover, and population density. Trans Am. Fish. Soc. 100:423-238.

Bjornn, T.C. and D.W. Reiser. 1991. Habitat requirements of anadromous salmonids. Influences of forest and rangeland management on salmonid fishes and their habitats. Am. Fish. Soc. Special Publ. 19: 83-138.

Bohn, C. 1989. Management of winter soil temperatures to control streambank erosion. p. 69-71. In: R.E. Gresswell, B.A. Barton, J.L. Kershner. (eds.). Practical approaches to riparian resource management: an educational workshop. May 8-11, 1989, Billings, Montana. U.S. Bureau of Land Management, Billings, Montana.

BNF (Boise National Forest). 1993. Biological assessment of Bear Valley Basin livestock grazing allotments — Effects on Snake River Basin spring/summer chinook salmon, Boise National Forest, Boise, Id., unpublished.

Bovee, K.D. 1982. A guide to stream habitat analysis using the instream flow incremental methodology. Instream flow information paper: No. 12. FWS/OBS-82/26. Instream Flow and Aquatic Systems Group, Western Energy and Land Use Team, US Fish and Wildlife Service, Fort Collins, Colorado.

Bruns, D.A., G.W. Minshall, C.E. Cushing, J.T. Brock, and R.L. Vannote. 1984. Tributaries as modifiers of the river continuum concept: analysis by polar ordination and regression models. Arch. Hydrobiol. 99:208-220.

Bruns, D.A., G.W. Minshall, C.E. Cushing, J.T. Brock, and R.L. Vannote. 1984. Tributaries as modifiers of the river continuum concept: analysis by polar ordination and regression models. Arch. Hydrobiol. 99:208-220.

Burton, T.A., W.H. Clark, G.W. Harvey, and T.R. Maret. 1991. Development of sediment criteria for the protection and propagation of salmonid fishes. p. 142-144. In: Biological criteria: research and regulation, Proceedings of a symposium. Environmental Protection Agency, Office of Water (WH-586), Washington, D.C. 20460, EPA-440/5-91-005. July 1991.

Carling, D.A. 1984. Deposition of fine and coarse sand in an open-work gravel bed. Can. J. Fish. Aquat. Sci. 41: 263-270.

Carson, M.A. and G.A. Griffiths. 1987. Bedload transport in gravel channels. J. Hydrol. 26(1):1-151.

Cederholm, C.J. and Scarlett, W.J., 1982. Seasonal immigrations of juvenile salmonids into four small tributaries of the Clearwater River, Washington, 1977-1981. Proceedings of the salmon and trout migratory behavior symposium, June 3-5, 1981. School of Fisheries, Univ. of Wash., Seattle, Wash.

Chapman, D.W., and K.P. McLeod. 1987. Development of criteria for sediment in the Northern Rockies ecoregion. EPA 910/9-87-162. USEPA, Region 10, Seattle, WA. April 1987. 279 p.

Clearwater National Forest. 1993. Clearwater National Forest monitoring and evaluation report, fiscal year 1992. Clearwater National Forest, Orofino, Id., unpublished.

Cooper, S.D. and L. Barmuta. 1993. Field experiments in biomonitoring. p. 399-441. In: D.M. Rosenberg and V.H. Resh (eds.). Freshwater biomonitoring and benthic macroinvertebrates, Chapman and Hall, New York, NY. 488 p.

Costanza, R. 1992. Toward an operational definition of ecosystem. p. 239-256. In: R. Costanza, B.G. Norton, and B.D. Haskell. 1992. Ecosystem health. New goals for environmental management. Island Press, Washington, D.C.

Coutant, C.C. 1995. Lost ecological carrying capacity in the mainstem. Workshop on ecological carrying capacity for Columbia Basin salmon habitats, September 6-7, 1995, Portland, Oregon. Battelle Northwest Laboratories.

Culp, J.M., F.J. Wrona, and R.W. Davies. 1986. Response of stream benthos and drift to fine sediment deposition versus transport. Can. J. Zool. 64:1345-1351.

Cummins, K.W. 1994. Bioassessment and analysis of functional organization of running water ecosystems. p. 155-169. In: S.L. Loeb and A. Spacie (ed.). Biological monitoring of aquatic systems. Lewis Publishers, Boca Raton, Florida. 381 p.

Cummins, K.W., G.L. Spengler, G.M. Ward, R.M. Speaker, R.W. Ovink, D.C. Mahan, and R.L. Mattingly. 1980. Processing of confined and naturally entrained leaf litter in a woodland stream ecosystem. Limnol. Oceanogr. 25(5):952-957.

Cummins, K.W., M.A. Wilzbach, D.M. Gates, J.B. Perry, and W.B. Taliaferro. 1989. Shredders and riparian vegetation. BioScience 39:24-30.

Cupp, C.E. 1989. Valley segment type classification for forested lands of Washington. Timber/Fish/Wildlife Ambient Monitoring Program. Washington Department of Natural Resources, Olympia, Washington. TFW-AM-89-001.

Cushing, C.E., C.D. McIntire, K.W. Cummins, G.W. Minshall, R.C. Petersen, J.R. Sedell, and R.L. Vannote. 1983. Relationships among chemical, physical, and biological indices along river continua based on multivariate analyses. Arch. Hydrobiol. 98(3):317-326.

Detenbeck, N.E., P.W. DeVore, G.J. Nieme, and A. Lima. 1992. Recovery of temperate-stream fish communities from disturbance: a review of case studies and synthesis of theory. Environmental Management 16(1):33-53.

Diplas, P. 1991. Interaction of fines with a gravel bed. p.5-10 to 5-16. In: Proceedings of the fifth federal interagency sedimentation conference, March 18-21, 1991, Las Vegas, Nevada. Interagency Advisory Committee on Water Data, Subcommittee on Sedimentation. Federal Energy Regulatory Commission.

Diplas, P. and J. Fripp. 1991. Bed material sampling: issues and answers. p. 2-81 to 2-88. In: Proceedings of the fifth federal interagency sedimentation conference, March 18-21, 1991, Las Vegas, Nevada. Interagency Advisory Committee on Water Data, Subcommittee on Sedimentation. Federal Energy Regulatory Commission.

Dolloff, C.A., D.G. Hankin, and G.H. Reeves. 1993. Basinwide estimation of habitat and fish populations in streams. Gen. Tech. Rep. SE-83. Asheville, NC. US Department of Agriculture, Forest Service, Southeastern Forest Experiment Station. 25 p.

Drury, W.H. and I.C.T. Nisbet. 1971. Inter-relations between developmental models in geomorphology, plant ecology, and animal ecology. p. 57-68. In: L. von Bertalannfy and A. Rapoport (eds.). Yearbook, Soc. Gen. Syst. Res.

Edington, J.M. and A.H. Hildrew. 1973. Experimental observations relating to the distribution of net-spinning Trichoptera in streams. Verh. Internat. Verein. Limnol. 18:1549-1558.

Elliott, J.M. 1977. Some methods for the statistical analysis of samples of benthic invertebrates. Freshwater Biological Association, Scientific Publication No. 25. Ambleside, England. 160 p.

EPA. 1993. Evaluation of draft technical guidance on biological criteria for streams and small rivers. EPA-SAB-EPEC-94-003. Prepared by the Science Advisory Board, Biological Criteria Subcommittee of the Ecological Processes and Effects Committee. 24 p.

Espinosa, F.A., Jr. 1991. Monitoring and evaluation report. USDA Forest Service, Clearwater National Forest, Region 1, 128 p.

Espinosa, F.A., Jr. 1992. Monitoring and evaluation report. USDA Forest Service, Clearwater National Forest, Region 1, 171 p.

Everest, F.H., F.B. Lotspeich, and W.R. Meehan. 1982. New perspectives on sampling, analysis, and interpretation of spawning gravel quality. p. 325-333. In Armantrout, N.B. (ed.). Acquisition and utilization of aquatic habitat inventory information. Proceedings of a symposium held 28-30, October, 1981, Hilton Hotel, Portland, Oregon. Western Division, American Fisheries Society.

Everest, F.H., R.L. Beschta, J.C. Scrivener, K.V. Koski, J.R. Sedell, and C.J. Cederholm. 1987. Fine sediment and salmonid production: a paradox. p. 98-142. In: E.O. Salo and T.W. Cundy, editors. Streamside management: forestry and fishery interactions. College of Forest Resources, University of Washington, Seattle. Contribution No. 57. Proceedings of a Symposium held at University of Washington, February 12-14, 1986.

Everest, F.H., F.B. Lotspeich, and W.R. Meehan. 1981. New perspectives on sampling, analysis, and interpretation of spawning gravel quality. Symposium on acquisition and utilization of aquatic habitat inventory information.

Everest, F.H., G.H. Reeves, J.R. Sedell, J. Wolfe, D. Hohler, and D.A. Heller. 1986. Abundance, behavior, and habitat utilization by coho salmon and steelhead trout in Fish Creek, Oregon, as influenced by habitat enhancement. Portland, OR: U.S. Department of Energy, Bonneville Power Administration. 100 p.

Everest, F.H., N.B. Armantrout, S.M. Keller, W.D. Parante, J.R. Sedell, T.E. Nickelson, J.N. Johnson, and G.N. Haugen. 1985. Salmonids. p. 200-230. In: Management of wildlife and fish habitats in western Oregon and Washington. US Forest Service, Pacific Northwest Region, Portland, Oregon.

Everhart, W.H. and W.D. Youngs. 1988. Principles of fishery science. Cornell University Press. 343 p.

Fausch, K.D., C.L. Hawkes, and M.G. Parsons. 1988. Models that predict standing crop of stream fish from habitat variables: 1950-85. U.S. Department of Agriculture, Forest Service, General Technical Report PNW-GTR-213, Pacific Northwest Research Station, Portland, Oregon. 52 p.

Fausch, K.D., J. Lyons, J.R. Karr, and P.L. Angermeier. 1990. Fish communities as indicators of environmental degradation. Am. Fisheries Soc. Symp. 8:123-144.

Fausch, K.D., J.R. Karr, and P.R. Yant. 1984. Regional application of an index of biotic integrity based on stream fish communities. Trans. Am. Fish. Soc. 113:39-55.

Finger, T. 1979. Patterns of interactive segregation in three species of sulpins (Cottus) in western Oregon. Ph.D. thesis. Oregon State University, Corvallis, Oregon. 126 p.

Fore, L.S., J.R. Karr, and R.W. Wisseman. 1995. A benthic index of biotic integrity for streams in the Pacific Northwest. Draft submitted to J. North Am. Benthological Society.

Fore, L.S., J.R. Karr, and L.L. Conquest. 1994. Statistical properties of an index of biological integrity used to evaluate water resources. Can. J. Fish. Aquat. Sci. 51:1077-1087.

Freeman, M.C., M.K. Crawford, J.C. Barrett, D.E. Facey, M.G. Flood, J. Hill, D.J. Stouder, and G.D. Grossman. 1988. Fish assemblage stability in a southern Appalachian stream. Can. J. Aquat. Sci. 45:1949-1958.

Frissell, C.A., 1992. Cumulative effects of land use on salmon habitat in southwest Oregon coastal streams. Unpub. Ph.D. thesis, Oregon State University, Corvallis, Oregon. 227 p.

Frissell, C.A., W.J. Liss, C.E. Warren, and M.D. Hurley. 1986. A hierarchical framework for stream habitat classification: viewing streams in a watershed context. Environmental Management 10:199-214.

Frissell, C.A., R.K. Nawa, and W.J. Liss. 1992. Water temperature and distribution and diversity of salmonid fishes in Sixes River Basin, Oregon, USA: Changes since 1965-1972. Chapter 4. In: C.A. Frissell. 1992. Cumulative effects of land use on salmon habitat in southwest Oregon coastal streams. Unpub. Ph.D. thesis, Oregon State University, Corvallis, Oregon. 227 p.

Fulton, L. 1968. Spawning areas and abundance of chinook salmon (*Oncorhynchus tshawytscha*) in the Columbia River basin—past and present. Special Scientific Report-Fisheries 571.

Gallant, A.L., T.R. Whittier, D.P. Larsen, J.M. Omernik, and R.M. Hughes. 1989. Regionalization as a tool for managing environmental resources. NSI Technology Services Corporation and U.S.E.P.A. Environmental Research Laboratory, Corvallis, Oregon. EPA/600/3-89/060. 152 p.

Gard, R. and G.A. Flittner. 1974. Distribution and abundance of fishes in Sagehen Creek, California. J. Wildlife Management 38:347-358.

Gebhardt, K.A., C. Bohn, S. Jensen, and W.S. Platts. 1989. Use of hydrology in riparian classification. p. 53-59. In: R.E. Gresswell, B.A. Barton, J.L. Kershner. (eds.). Practical approaches to riparian resource management: an educational workshop. May 8-11, 1989, Billings, Montana. U.S. Bureau of Land Management, Billings, Montana.

Gorman, D.T. and J.R. Karr. 1978. Habitat structure and stream fish communities. Ecology 59:507-515.

Gorte, R.W. 1994. Forest Service reform: concerns and considerations. Testimony before the House Committee on Natural Resources, Subcommittees on Oversight and Investigations and on National Parks, Forests, and Public Lands. Office of Technology Assessment, Congress of the United States, 9p.

Gowan, C., M.K. Young, K.D. Fausch, and S.C. Riley. 1994 .Restricted movement in resident stream salmonids: a paradigm lost? Can. J. Fish. Aquat. Sci. 51:2626-2637.

Grant, G.E. 1986. Downstream effects of timber harvest activities on the channel and valley floor morphology of western Cascade streams. Ph.D dissertation. Johns Hopkins University. Baltimore, Maryland. 363 p.

Grant, G. 1988. The RAPID technique: a new method for evaluating downstream effects of forest practices on riparian zones. U.S.D.A. Forest Service General Technical Report PNW-GTR-220, Pacific Northwest Research Station, Portland, Oregon. 36p.

Grossman, G.D., J.F. Dowd, and M. Crawford. 1990. Assemblage stability in stream fishes: a review. Environmental Management 14(5):661-671.

Grost, R.T., W.A. Hubert, and T.A. Wesche. 1991a. Field comparison of three devices used to sample substrate in small streams. N. Am. J. Fish. Management 11(3):347-353.

Grost, R.T., W.A. Hubert, and T.A. Wesche. 1991b. Description of brown trout redds in a mountain stream. Trans. Am. Fish. Soc. 120:582-588.

Hagans, D.K. and W.E. Weaver. 1987. Magnitude, cause and basin response to fluvial erosion, Redwood Creek basin, northern California. p. 419-428. In: Erosion and sedimentation in the Pacific Rim, Proceedings of the Corvallis Symposium, August, 1987. IAHS Publ. No. 165.

Hankin, D.G. 1984. Multistage sampling designs in fisheries research: applications in small streams. Can. J. Fish. Aq. Sci. 41:1575-1591.

Hankin, D.G. and G.H. Reeves. 1988. Estimating total fish abundance and total habitat area in small streams based on visual estimation methods. Can. J. Fish. Aq. Sci. 45:834-844.

Hawkes, H.A. 1977. Biological classification of rivers. Conceptual basis and ecological validity. In: J.S. Alabaster (ed.). Biological Monitoring of Inland Fisheries. Applied Science Publishers, Ltd. London, England.

Hawkes, H.A. 1975. River zonation and classification, p. 312-374. In: B.A. Whitton (ed.). U. Cal. Press.

Hawkins, C.P. 1985. Substrate associations and longitudinal distributions in species of Ephemerellidae (Epermeroptera: Insecta) from western Oregon. Freshwater Invertebrate Biology 4:181-188.

Hawkins, C.P., M.L. Murphy, and N.H. Anderson. 1982. Effects of canopy, substrate composiition, and gradient on the structure of macroinvertebrate communities in Cascade Range streams of Oregon. Ecology 63(6):1840-1856.

Hawkins, C.P., Kershner, J.L., Bisson, P.A., Bryant, M.D., Decker, L.M., Gregory, S.V., McCullough, D.A., Overton, C.K., Reeves, G.H., Steedman, R.J., and Young, M.K. 1993. A hierarchical approach to classifying stream habitat features. Fisheries 18: 3-12.

Heede, B.H. 1975. Mountain watersheds and dynamic equilibrium. p. 407-420. Proc.of Watershed Management Symposium. August 11-13, 1975, Logan, Utah. ASCE, Irrigation and Drainage Division.

Heede, B.H. 1980. Stream dynamics: an overview for land managers. Gen. Tech. Report RM-72, USDA Forest Service, Rocky Mountain Forest and Range Experiment Station, Fort Collins, Colorado. 26 p.

Heerdegen, R.G. and M.A. Beran. 1982. Quantifying source areas through land surface curvature and shape. J. Hydrol. 57:359-373.

Henjum, M.G., J.R. Karr, D.L. Bottom, D.A. Perry, J.C. Bednarz, S.G. Wright, S.A. Beckwitt, and E. Beckwitt. 1994. Interim protection for late-successional forests, fisheries, and watersheds: National forests east of the Cascade crest, Oregon and Washington. The Wildlife Society, Bethesda, MD.

Herrington, R.B. and D.K. Dunham. 1967. A technique for sampling general fish habitat characteristics of streams. USDA Forest Service. Res. Paper INT-41. 12 p. Intermountain Research Station. Ogden, Utah.

Hill, M.T., W.S. Platts, and R.L. Beschta. 1991. Ecological and geomorphological concepts for instream and out-of-channel flow requirements. Rivers 2(3):198-210.

Hogan, D.L. 1987. The influence of large organic debris on channel recovery in the Queen Charlotte Islands, British Columbia, Canada. p. 343-353. In: Erosion and sedimentation in the Pacific Rim, Proceedings of the Corvallis Symposium, August, 1987. IAHS Publ. No. 165.

Holtby, L.B. 1988. Effects of logging on stream temperatures in Carnation Creek, British Columbia, and associated impacts on the coho salmon (*Onchorhynchus kisutch*). Can. J. Fish. Aquat. Sci. 45:502-515.

Horwitz, R.J. 1978. Temporal variability patterns and the distributional patterns of stream fishes. Ecol. Monogr. 48:307-321.

House, R. 1995. Temporal variation in abundance of an isolated population of cutthroat trout in western Oregon, 1981-1991. N. Am. J. Fish. Management 15:33-41.

Huet, M. 1959. Profiles and biology of western European streams as related to fish management. Trans. Am. Fish. Soc. 88(1):155-163.

Hughes, R.M. and J.M. Omernik. 1982. A proposed approach to determine regional patterns in aquatic ecosystems. p. 92-102. In: N.B. Armantrout, (ed.). Acquisition and utilization of aquatic habitat inventory information. Proceedings of a symposium held 28-30, October, 1981, Hilton Hotel, Portland, Oregon. Western Division, American Fisheries Society.

Hughes, R.M. and J.R. Gammon. 1987. Longitudinal changes in fish assemblages and water quality in the Willamette River, Oregon. Trans. Am. Fish. Soc. 116:196-209.

Hughes, R.M., E. Rexstad, and C.E. Bond. 1987. The relationship of aquatic ecoregions, river basins, and icthyogeographic regions of Oregon. Copeia 2:423-432.

Hughes, R.M., T.R. Whittier, C.M. Rohm, and D.P. Larsen. 1990. A regional framework for establishing recovery criteria. Environmental Management 14(5):673-683.

Huntington, C.W. 1994. Fish habitat and salmonid abundance within managed and unroaded landscapes on the Clearwater National Forest, Idaho. Prepared for Eastside Ecosystem Management Project, USDA Forest Service, Walla Walla, WA. Order No. 43-0E00-4-9106. 40 p. + appendices.

Johnson, J.H. and P.A. Kucera. 1985. Summer-autumn habitat utilization of subyearling steelhead trout in tributaries of the Clearwater River, Idaho. Can. J. Zool. 63:2283-2290.

Johnson, S.W., J. Heifetz, and K.V. Koski. 1986. Effects of logging on the abundance and seasonal distribution of juvenile steelhead in some southeastern Alaska streams. North American Journal of Fisheries Management 6:532-537.

Jowett, J.G. 1990. Factors related to the distribution and abundance of brown and abundance of brown and rainbow trout in New Zealand clear-water rivers. N.Z. J. Mar. Freshwat. Res. 24(3):429-440.

Jowett, I.G. and M.J. Duncan. 1990. Flow variability in New Zealand rivers and its relationship to in-stream habitat and biota. N.Z. J. Mar. Freshwater Research 24(3):305-317.

Kappesser, G.B. 1993. Riffle stability index. A procedure to evaluate stream reach and watershed equilibrium. Idaho Panhandle NF, 10 p.

Karr, J.R. 1981. Assessment of biotic integrity using fish communities. Fisheries 6(6):21-27.

Karr, J.R. 1990. Biological integrity and the goal of environmental legislation: lessons for conservation biology. Conservation Biology 4(3):244-250.

Karr, J.R. 1991. Biological integrity: a long neglected aspect of water resource management. Ecological Applications 1(1):66-84.

Karr, J.R. 1994. Restoring wild salmon: we must do better. Illahee 10(4):316-319.

Karr, J.R. 1995. Protecting aquatic ecosystems: clean water is not enough. p. 7-13. In: W.S. Davis and T.P. Simon. Biological assessment and criteria. Tools for water resource planning and decision making. Lewis Publishers, Boca Raton.

Karr, J.R. 1995. Clean water is not enough. Illahee 11(1-2):51-61.

Karr, J.R. and I.J. Schlosser. 1978. Water resources and the land-water interface. Science 201:229-234.

Karr, J.R. and D.R. Dudley. 1981. Ecological perspective on water quality goals. Environmental Management 5(1):55-68.

Kelson, K.I. and S.G. Wells. 1989. Geologic influences on fluvial hydrology and bedload transport in small mountainous watersheds, Northern New Mexico, U.S.A. Earth Surface Processes and Landforms 14:671-690.

Kerans, B.L. and J.R. Karr. 1994. A benthic index of biotic integrity (B-IBI) for rivers of the Tennessee Valley. Ecological Applications 4(4):768-785.

Ketcheson, G.L. 1986. Sediment rating equations: an evaluation for streams in the Idaho batholith. General Technical Report INT-213. USDA Forest Service, Intermountain Research Station, Ogden, Utah. 12 p.

Ketcheson, G.L. and W.F. Megahan. 1991. Sediment tracing in step-pool granitic streams in Idaho. In: Proceedings of the fifth federal interagency sedimentation conference, March 18-21, 1991, Las Vegas, Nevada. Interagency Advisory Committee on Water Data, Subcommittee on Sedimentation. Federal Energy Regulatory Commission.

Kiefer, R.B. and K.A. Forster. 1991. Intensive evaluation and monitoring of chinook salmon and steelhead trout production, Crooked River and Upper Salmon River sites. Idaho Department of Fish and Game, Annual Report to Bonneville Power Administration, Portland, Oregon.

King, J.G. and R.F Thurow. 1991. Progress report: Sediment monitoring techniques validation. USFS Intermountain Research Station, Boise, Id., unpublished.

Kondolf, G.M. and M. Larson. 1995. Historical channel analysis and its application to riparian and aquatic habitat restoration. Aquatic Conservation: Marine and Freshwater Ecosystems 5:109-126.

Kovalchik, B.L. 1987. Riparian zone associations. Deschutes, Ochoco, Fremont, and Winema National Forests. USDA Forest Service. Pacific Northwest Region. R6 ECOL TP-279-87. 171 p.

Kovalchik, B.L. and L.A. Chitwood. 1988. The use of geomorphology in the classification of riparian plant associations in mountainous landscapes of central Oregon, USA. Paper presented at The International Forested Wetlands Resource: Identification and Inventory, Sept. 19-22, 1988. Baton Rouge, Louisiana, USA.

Kuehne, R.A. 1962. A classification of streams illustrated by fish distributed in an eastern Kentucky Creek. Ecology 43(4):608-614.

Larsen, D.P., J.M. Omernik, R.M. Hughes, C.M. Rohm, T.R. Whittier, A.J. Kinney, A.L. Gallant, and D.R. Dudley. 1986. Correspondence between spatial patterns in fish assemblages in Ohio streams and aquatic ecoregions. Env. Management 10(6):815-828.

Lawler, D.M. 1993. The measurement of river bank erosion and lateral channel change: a review. Earth Surface Processes and Landforms 18:777-821.

Lee, M.T. and J.W. Delleur. 1976. A variable source area model of the rainfall-runoff process based on the watershed stream network. Water Resour. Res. 12(5):1029-1036.

Lenat, D.R. and M.T. Barbour. 1994. Using benthic macroinvertebrate community structure for rapid, cost-effective, water quality monitoring: rapid bioassessment. p. 187-215. In: S.L. Loeb and A. Spacie (ed.). Biological monitoring of aquatic systems. Lewis Publishers, Boca Raton, Florida. 381 p.

Li, H.W., C.B. Schreck, C.E. Bond, and E. Rexstad. 1987. Factors influencing changes in fish assemblages of Pacific Northwest streams. In: W.J. Matthews and D.C. Heins (eds.). Community and evolutionary ecology of North American stream fishes. Univ. of Oklahoma Press. Norman and London.

Li, H.W., G.A. Lamberti, T.N. Pearsons, C.K. Tait, J.L. Li, and J.C. Buckhouse. 1994. Cumulative effects of riparian disturbances along high desert trout streams of the John Day Basin, Oregon. Trans. Am. Fish. Soc. 123:627-640.

Lisle, T.E. 1989. Using 'residual depths' to monitor pool depths independently of discharge. Research Note PSW-394, USDA Forest Service PSW Station, Berkeley, CA. 4 p.

Lisle, T.E. and R.E. Eads. 1991.Methods to measure sedimentation of spawning gravels. US Forest Service, Pacific Southwest Research Station. Research Note PSW-411. 7 p.

Lisle, T. and S. Hilton. 1992. Fine sediment in pools: an index of how sediment is affecting a stream channel. FHR Currents. R-5 Fish Habitat Relationship Tech. Bull. 6 p.

Ludwig, D. 1996. The end of the beginning. Ecological Applications 6(1):16-17.

Lyons, J. 1989. Correspondence between the distribution of fish assemblages in Wisconsin streams and Omernik's ecoregions. Am. Midl. Nat. 122:163-182.

Lyons, J.K., and R.L. Beschta. 1983. Land use, floods, and channel changes: upper Middle Fork Willamette River, Oregon (1936-1980). Water Resources Research 19:463-471.

MacDonald, L.H., A.W. Smart, R.C. Wissmar. 1991. Monitoring guidelines to evaluate effects of forestry activities on streams in the Pacific Northwest and Alaska. EPA 910/9-91-001. USEPA, Region 10, Seattle, WA. 166 p.

Magdych, W.P. 1984. Salinity stresses along a complex river continuum: effects on mayfly (Ephemeroptera) distributions. Ecology 65:1662-1672.

Maret, T.R., T.A. Burton, G.W. Harvey, and W.H. Clark. 1993. Field testing of new monitoring protocols to assess brown trout spawning habitat in an Idaho stream. N. Am. J. Fisheries Management 13:567-580.

Marschall, E.A. and L.B. Crowder. 1996. Assessing population responses to multiple anthropogenic effects: a case study with brook trout. Ecological Applications 6(1):152-167.

Mattax, B.L. and T.M. Quigley. 1989. Validation and sensitivity analysis of the stream network temperature model on small watersheds in northeast Oregon. p. 391-400. In: W.W. Woessner, and D.F. Potts (eds.). Am. Water Resour. Ass., Proc. of the Symp. on Headwaters Hydrology.

McCullough, D.A. 1988. A systems classification of watersheds and streams. Technical Report 88-3. Columbia River Inter-Tribal Fish Commission. 217 p.

McCullough, D.A. 1990. Classification of streams within a landscape perspective. Coordinated Information System Annual Report. System monitoring and evaluation project (88-108). Columbia River Inter-Tribal Fish Commission, Portland, Oregon.

McDonald, L.H., A.W. Smart, and R.C. Wissmar. 1991. Monitoring guidelines to evaluate effects of forestry activities on streams in the Pacific Northwest and Alaska. US Environmental Protection Agency. EPA 910/9-91-001. Center for Streamside Studies in Forestry, Fisheries and Wildlife, University of Washington, Seattle, Washington.

McElravy, E.P., G.A. Lamberti, and V.H. Resh. 1989. Year-to-year variation in the aquatic macroinvertebrate fauna of a northern California stream. J. N. Am. Benthol. Soc. 8(1):51-63.

McIntosh, B.A., D.M. Price, C.E. Torgersen, and H.W. Li. 1995. Distribution, habitat utilization, movement patterns, and the use of thermal refugia by spring chinook in the Grande Ronde, Imnaha, and John Day Basins. Progress report to the Bonneville Power Administration, Project No. 88-108, FY-1995.

McIntosh, B.A., J.R. Sedell, J.E. Smith, R.C. Wissmar, S.E. Clarke, G.H. Reeves, and L.A. Brown. 1994. Management history of eastside ecosystems: changes in fish habitat over 50 years, 1935 to 1992. Gen. Tech. Report PNW-GTR-321. USDA Forest Service. Pacific Northwest Forest and Range Experiment Station. Portland, Oregon. 54 p.

McMahon, T.E. and G.F. Hartman. 1989. Influence of cover complexity and current velocity on winter habitat use by juvenile coho salmon (*Oncorhynchus kisutch*). Can. J. Fish. Aquat. Sci. 46:1551-1557.

Meffe, G.K. and A.L. Sheldon. 1990. Post-defaunation recovery of fish assemblages in southeastern blackwater streams. Ecol. 71(2):657-667.

Meffe, G.K., C.R. Carroll. 1994. Principles of conservation biology. Sinauer Associates, Inc., Publishers, Sunderland, Massachussetts.

Megahan, W.F. N.F. Day, and T.M. Bliss. 1978. Landslide occurrence in the western and central northern Rocky Mountain physiographic province in Idaho. p. 116-139. Proceedings: Fifth N. Amer. Forest Soils Conference, Colorado State University, Fort Collins, Colorado.

Miller, R.J. and E.L. Brannon. 1982. The origin and development of life history patterns in Pacific salmonids. p. 296-309. In: E.L. Brannon and E.O. Salo (eds.). Proceedings of the salmon and trout migratory behavior symposium. June 3-5, 1981, School of Fisheries, University of Washington, Seattle, Washington.

Mills, L.S., M.E. Soule, and D.F. Doak. 1993. The keystone-species concept in ecology and conservation. BioScience 43(4):219-224.

Minshall, G.W. 1993. Stream-riparian ecosystems: rationale and methods for basin-level assessments of management effects. In: M.E. Jensen and P.S. Bourgeron (eds.). 1993 Eastside forest ecosystem health assessment. Volume II: Ecosystem management: Principles and applications. US Forest Service, Pacific Northwest Research Station, Portland, Oregon.

Moore, K.M.S. and S.V. Gregory. 1988a. Response of young-of-the-year cutthroat trout to manipulation of habitat structure in a small stream. Trans. Am. Fish. Soc. 117:162-170.

Moore, K.M.S. and S.V. Gregory. 1988b. Summer habitat utilization and ecology of cutthroat trout fry (*Salmo clarki*) in Cascade Mountain streams. Can. J. Fish. Aquat. Sci. 45:1921-1930.

Morisawa, M. 1981. Fluvial geomorphology. Allen and Unpin. London.

Moyle, P.B. and D.M. Baltz. 1985. Microhabitat use by an assemblage of California stream fishes: developing criteria for instream flow determinations. Trans. Am. Fish. Soc. 114:695-704.

Moyle, P.B. and B. Vondracek. 1985. Persistence and structure of the fish assemblage in a small California stream. Ecology 66(1):1-13.

Murphy, M.L. and J.D. Hall. 1981. Varied effects of clear-cut logging on predators and their habitat in small streams of the Cascade Mountains, Oregon. Can. J. Fish. Aquat. Sci. 38:137-145.

Murphy, M.L., C.P. Hawkins, and N.H. Anderson. 1981. Effects of canopy modification and accumulated sediment on stream communities. Trans. Am. Fish. Soc. 110(4):469-478.

Murphy, M.L., J. Heifetz, S.W. Johnson, K.V. Koski, and J.F. Thedinga. 1986. Effects of clear-cut logging with and without buffer strips on juvenile salmonids in Alaskan streams. Can. J. Fish. Aquat. Sci. 43:1521-1533.

Nawa, R.K., C.A. Frissell, and W.J. Liss. 1988. Life history and persistence of anadromous fish stocks in relation to stream habitats and watershed classification. Annual Progress Report. Prepared for Oregon Department of Fish and Wildlife, Portland, Oregon. 37 p.

Nelson, R.L., W.S. Platts, D.P. Larsen. and S.E. Hensen. 1992. Trout distribution and habitat in relation to geology and geomorphology in the North Fork Humboldt River drainage, northeastern Nevada. Trans. Am. Fisheries. Soc. 121:405-426.

NMFS (National Marine Fisheries Service). 1995. Draft biological opinion. Thunderbolt Wildlife Recovery Project, Endangered Species Act, Section 7 Consultation, conducted by NMFS with the USFS, Boise and Payette National Forests. September 22, 1995.

Norris, R.H. and A. Georges. 1993. Analysis and interpretation of benthic macroinvertebrate surveys. p. 234-286. In: D.M. Rosenberg and V.H. Resh (eds.). Freshwater biomonitoring and benthic macroinvertebrates, Chapman and Hall, New York, NY. 488 p.

Northcote, T.G. and D.W. Wilkie. 1963. Underwater census of stream fish populations. Trans. Am. Fish. Soc. 92: 146-151.

O'Neill, D.L. DeAngelis, J.B. Waide, and T.F.H. Allen. 1986. A hierarchical concept of ecosystems. Monographs in Population Biology 23. Princeton University Press, Princeton. 253 p.

Omernik, J.M. 1987. Ecoregions of the conterminous United States (text and map). Annals of the Association of Am. Geogr. 77(1):118-125.

Omernik, J.M. and A.L. Gallant. 1986. Ecoregions of the Pacific Northwest. EPA/600/3-86/033. U.S.Environmental Protection Agency, Environmental Research Laboratory, Corvallis, Oregon. 39 p.

O'Neill, R.V., J.R. Kahn, J.R. Duncan, S. Elliott, R. Efroymson, H. Cardwell, and D.W. Jones. 1996. Ecological Applications 6(1):23-24.

Orsborn, J.F. 1980. Estimating streamflow characteristics at spawning sites in Oregon. CERL-051. US Environmental Protection Agency, Corvallis, Oregon.

Orsborn, J.F. 1990. Quantitative modeling of the relationships among basin, channel and habitat characteristics for classification and impact assessment. Timber/Fish/Wildlife Program. CMER Project 16D.

Overton, C.K., M.A. Radko, and R.L. Nelson. 1993. Fish habitat conditions: using the Northern/Intermountain Regions' inventory procedures for detecting differences on two differently managed watersheds. Gen. Tech. Rep. INT-300. USDA Forest Service. Intermountain Research Station. Ogden, Utah. 14 p.

Paller, M.H. 1995. Interreplicate variance and statistical power of electrofishing data from low-gradient streams in the southeastern United States. N. Am. J. Fish. Management 15:542-550.

Parizek, R.R. 1978. In: D.R. Coates and J.D. Vitek (eds). Thresholds in geomorphology. George Allen and Unpin. London. 498 p.

Patten, R. 1989. WATBAL, Watershed response model for forest management. Technical user guide, Clearwater National Forest. 25 p.

Pearson, R.G. and N.V. Jones. 1984. The River Hull, a northern English chalk stream: the zonation of the macro-invertebrate fauna with reference to physical and chemical features.
Arch. Hydrobiol. 100(2):137-157.

Pearsons, T.N., H.W. Li, and G.A. Lamberti. 1992. Influence of habitat complexity on resistance to flooding and resilience of stream fish assemblages. Trans. Am. Fish. Soc. 121:427-436.

Peterson, J.T. and C.F. Rabeni. 1995. Optimizing sampling effort for sampling warmwater stream fish communities. N. Am. J. Fish. Management 15:528-541.

Peterson, R.H. and L. Vaneeckhaute. 1992. Distributions of Ephemeroptera, Plectoptera, and Trichoptera of three maritime catchments differing in pH. Freshwater Biol. 27:65-78.

Pickup, G., G.N. Bastin, and V.H. Chewings. 1994. Remote-sensing-based condition assessment for nonequilibrium rangelands under large-scale commercial grazing. Ecological Applications 4(3):497-517.

Pimm, S.L. 1991. The balance of nature? Ecological issues in the conservation of species and communities. The University of Chicago Press, Chicago. 434 p.

Plafkin, J.L., M.t. Barbour, K.D. Porter, S.K. Gross, and R.M. Hughes. 1989. Rapid bioassessment protocols for use in streams and rivers. Benthic macroinvertebrates and fish. EPA/444/4-89/001. Office of Water Regulations and Standards, US Environmental Protection Agency, Washington, DC.

Platts, W.S. 1974. Geomorphic and aquatic conditions influencing salmonids and stream classification. Surface Environment and Mining Program. 199 p.

Platts, W.S., W.H. Megahan, and G.W. Minshall. 1983. Methods for evaluating stream, riparian, and biotic conditions. Gen. Tech. Report INT-138. USFS Intermountain Forest and Range Experiment Station, Ogden, UT. 70 p.

Platts, W.S., Armour, C., Booth, G.D., Bryant, M., Bufford, J.L., Culpin, P., Jensen, S., Lienkaemper, G.W., Minshall, G.W., Monsen, S.B., Nelson, R.L., Sedell, J.R., and Tuhy, J.S. 1987. Methods for evaluating riparian habitats with applications to management. USFS Gen. Tech. Report INT-221, Intermountain Research Station, Ogden, Utah.

Platts, W.S., R.J. Torquemada, M.L. McHenry, and C.K. Graham. 1989. Changes in salmon spawning and rearing habitat from increased delivery of fine sediment to the South Fork Salmon River, Idaho. Trans. Am. Fish. Soc. 118:274-283.

Poff, N.L. and J.D. Allan. 1995. Functional organization of stream fish assemblages in relation to hydrological variability. Ecology 76(2):606-627.

Rabeni, C.F. and G.W. Minshall. 1977. Factors affecting microdistribution of stream benthic insects. Oikos 29:33-43.

Rahel, F.J. and W.A. Hubert. 1991. Fish assemblages and habitat gradients in a Rocky Mountain-Great Plains stream: biotic zonation and additive patterns of community change. Trans. Am. Fish. Soc. 120:319-332.

Ralph, S.C., G.C. Poole, L.L. Conquest, and R.J. Naiman. 1994. Stream channel morphology and woody debris in logged and unlogged basins of western Washington. Can. J. Fish. Aquat. Sci. 51:37-51.

Reeves, G.H., F.H. Everest, and J.D. Hall. 1987. Interaction between redside shiner (*Richarsonius balteatus*) and the steelhead trout (*Salmo gairdneri*) in western Oregon: the influence of water temperature. Can. J. Fisheries and Aquatic Sciences 44:1603-1613.

Reeves, G.H., Everest, F.H., Sedell, J.R., and Hohler, D.B., 1990. Influence of habitat modifications on habitat composition and anadromous salmonid populations in Fish Creek, Oregon, 1983-1988. BPA Project No. 84-11, Bonneville Power Admin., Div. of Fish and Wildlife, Portland, Oregon.

Reeves, G.H., Everest, F.H., and Sedell, J.R., 1993. Diversity of juvenile anadromous salmonid assemblages in coastal Oregon Basins with different levels of timber harvest. Trans. Amer. Fish Soc. 122: 309-317.

Resh, V.H. and E.P. McElravy. 1993. Contemporary quantitative approaches to biomonitoring using benthic macroinvertebrates. In: D.M. Rosenberg and V.H. Resh (eds.). Freshwater biomonitoring and benthic macroinvertebrates, Chapman and Hall, New York, NY. 488 p.

Resh, V.H. and J.K. Jackson. 1993. Rapid assessment approaches to biomonitoring using benthic macroinvertebrates. p. 195-223. In: D.M. Rosenberg and V.H. Resh (eds.). Freshwater biomonitoring and benthic macroinvertebrates, Chapman and Hall, New York, NY. 488 p.

Rhodes, J.J. 1995. A comparison and evaluation of existing land management plans affecting spawning and rearing habitat of Snake River basin salmon species listed under the Endangered Species Act. NMFS/BIA Inter-Agency Agreement 40ABNF3. Submitted to National Marine Fisheries Service, Portland, Oregon. December 1994. Columbia River Inter-Tribal Fish Commission. 108 p.

Rhodes, J.J., D.A. McCullough, and F. A. Espinosa, Jr. 1994. A coarse screening process for potential application in ESA consultations. NMFS/BIA Inter-Agency Agreement 40ABNF3. Submitted to National Marine Fisheries Service, Portland, Oregon. December 1994. Columbia River Inter-Tribal Fish Commission. 127 p. + appendices.

Rice, R.M. 1992. The Science and politics of BMPs in forestry: California experiences. p. 385-400. In: R.J. Naiman (ed.). Watershed management. Balancing sustainability and environmental change. Springer-Verlag. New York. 542 p.

Ricker, W.E. 1934. An ecological classification of certain Ontario streams. Ontario Fish. Res. Lab. Publ. No. 49. Univ. Toronto Press. Biological Series No :1-114.

Ries, R.D. and D.C. Burns. 1989. Embeddedness of salmonid habitat of selected streams on the Payette National Forest 1987-1988. USFS, Payette National Forest. Unpublished report. 36 p.

Ringler, N.H. and J.D. Hall. 1975. Effects of logging on water temperature and dissolved oxygen in spawning beds. Trans. Am. Fish. Soc. 104:111-121.

Ringler, N.H. andJ.D. Hall. 1988. Vertical distribution of sediment and organic debris in coho salmon (*Oncorhynchus kisutch*) redds in three small Oregon streams. Can. J. Fish. Aquat. Sci. 45:742-747.

Riparian Habitat Subcommittee of the Oregon/Washington Interagency Wildlife Committee. 1979. Managing riparian ecosystems (zones) for fish and wildlife in eastern Oregon and eastern Washington. Oregon Department of Fish and Wildlife, US Forest Service, US Fish and Wildlife Service, US Bureau of Land Management, Soil Conservation Service, Washington Department of Game.

Rohm, C.M., J.W. Giese, and C.C. Bennett. 1987. Evaluation of an aquatic ecoregion classification of streams in Arkansas. J. Freshwater Ecol. 4:127-140.

Rood, S. and J.M. Mahoney. 1992. Instream flow needs for riparian vegetation: cottonwood forest ecosystems. p. 55-73. In: Instream flow needs. Seminar proceedings, Interdepartmental instream flow needs task force. April 14, 1992, Edmonton Inn. Alberta departments of Environment, Agriculture, Forestry, Lands and Wildlife, Water Resources Commission, Tourism, Parks and Recreation, and Municipal Affairs.

Rosenberg and V.H. Resh (eds.). Freshwater biomonitoring and benthic macroinvertebrates, Chapman and Hall, New York, NY. 488 p.

Rosgen, D.L. 1985. A stream classification system. p. 91-95. In: R.R. Johnson, C.D. Ziebell, D.R. Patton, P.F. Ffolliott, and R.H. Hamre, editors. Riparian ecosystems and their management: reconciling conflicting uses. First North American Riparian Conference, Tucson. U.S.D.A. Forest Service, General Technical Report RM-120, Fort Collins, Colorado. 523 p.

Rosgen, D.L. 1994. A classification of natural rivers. Catena 22:169-199.

Schill, D.J. and J.S. Griffith. 1984. Use of underwater observations to estimate cutthroat abundance in the Yellowstone River. N. Am. J. Fish. Manage. 4:479-487.

Schindler, D.W. 1987. Detecting ecosystem responses to anthropogenic stress. Can. J. Fish. Aquat. Sci. 44:6-25.

Schindler, D.W. 1996. The environment, carrying capacity and economic growth. Ecol. Applications 6(1):17-19.

Schlosser, I.J. 1982. Fish community structure and function along two habitat gradients in a headwater stream. Ecol. Monogr. 52:395-414.

Schlosser, I.J. 1985. Flow regime, juvenile abundance, and the assemblage structure of stream fishes. Ecology 66(5):1484-1490.

Schlosser, I.J. 1991. Stream fish ecology: a landscape perspective. BioScience 41(10):704-712.

Schumm, S.A. and R.W. Lichty. 1965. Time, space and causality in geomorphology. Am. J. Sci. 263:110-119.

Scrivener, J.C. and M.J. Brownlee. 1989. Effects of forest harvesting on spawning gravel and incubation survival of chum (*Oncorhynchus keta*) and coho salmon (*Oncorhynchus kisutch*) in Carnation Creek, British Columbia. Can. J. Fish. Aquat. Sci. 46:681-696.

Scully, R.J. and C.E. Petrosky. 1991. Idaho habitat and natural production Monitoring Part I. General Monitoring Subproject Annual Report 1989. BPA Project No. 83-7, Bonneville Power Admin., Div. of Fish and Wildlife, Portland, Or.

Seyedbagheri, K.A., M.L. McHenry, and W.S. Platts. 1987. An annotated bibliography of the hydrology and fishery studies of the South Fork of the Salmon River. USFS Gen. Tech. Report INT-235. Intermountain Research Station, Ogden, Utah.

Shaw, D.W. and G.W. Minshall. 1980. Colonization of an introduced substrate by stream macroinvertebrates. Oikos 34(3):259-271.

Simonson, T.D., J. Lyons, and P.D. Kanehl. 1994. Quantifying fish habitat in streams: transect spacing, sample size, and a proposed framework. N. Am. J. Fish. Management 14:607-615.

Skille, J. and J. King. 1988. Proposed cobble embeddedness methodology. Review draft. Idaho Department of Health and Welfare, Department of Environmental Quality, Boise, Idaho.

Statzner, B. 1988. Growth and Reynolds number of lotic macroinvertebrates: a problem for adaptation of shape to drag. Oikos 51:84-87.

Statzner, B. and B. Higler. 1986. Stream hydraulics as a major determinant of benthic invertebrate zonation patterns. Freshwater Biology 16:127-139.

Statzner, B. and R. Muller. 1989. Standard hemispheres as indicators of flow characteristics in lotic benthos research. Freshwater Biology 21:445-459.

Steedman, R.J. 1988. Modification and assessment of an index of biotic integrity to quantify stream quality in southern Ontario. Can. J. Fish. Aquat. Sci. 45:492-501.

Sullivan, K. 1986. Hydraulics and fish habitat in relation to channel morphology. Ph.D. Thesis. John Hopkins University. Baltimore, MD. 407 p.

Theurer, F.D., I. Lines, and T. Nelson. 1985. Interaction between riparian vegetation, water temperature, and salmonid habitat in the Tucannon River. Water Resources Bull 21(1):53-64.

Torquemada, R.J. and W.S. Platts. 1988. A comparison of sediment monitoring techniques of potential use in sediment/fish population relationships. Part III. USDA Forest Service. In: Idaho habitat evaluation for off-site mitigation record. Annual report 1987. US Department of Energy, Bonneville Power Administration, Division of Fish and Wildlife. Contract No. DE-A179-84BP13381, Project 83-7.

Townsend, C.R. and A.G. Hildrew. 1984. Longitudinal pattern in detritivore communities of acid streams: a consideration of alternative hypotheses. Verh. Int. Ver. Limnol. 22:1953-1958.

Tschaplinski, P.J. and G.F. Hartman. 1983. Winter distribution of juvenile coho salmon (*Oncorhynchus kisutch*) before and after logging in Carnation Creek, British Columbia, and some implications for overwinter survival. Can. J. Fish. Aquat. Sci. 40:452-461.

USFS. 1981. Guide for predicting sediment yields from forested watersheds. USDA, Forest Service, Northern and Intermountain Regions. 41p.

USFS. 1983. Guide for predicting salmonid response to sediment yields in Idaho batholith watersheds. USDA Forest Service Northwestern Regional Intermountain Region. 95 pp.

USFS. 1993. WATSED. A water and sediment prediction model. Northern Region, US Forest Service. 112 p.

USFS. 1994. USFS monitoring plan: Section 7 fish habitat monitoring protocol for the upper Columbia River Basin, Pacific Northwest Region, Intermountain Region, Northern Region. 6th revision. 61 p.

USFS, NMFS, USBLM, USFWS, USNPS, USEPA. 1993. Forest ecosystem management: An ecological, economic, and social assessment. USFS PNW Region, Portland, Oregon.

USFS and USBLM. 1994. Record of decision for amendments to Forest Service and Bureau of Land Management Planning documents within the range of the northern spotted owl/standards and guidelines for management of habitat for late successional and old-growth forest related species within the range of the northern spotted owl. Interagency Supplemental EIS Team, Portland, Oregon.

USFS and USBLM. 1995. Decision notice and environmental assessment for the interim strategies for managing anadromous fish-producing watersheds in eastern Oregon and Washington, Idaho, and portions of California. USFS and USBLM, Washington, DC.

Vannote, R.L., G.W. Minshall, K.W. Cummins, J.R. Sedell, and C.E. Cushing. 1980. The river continuum concept. Canadian Journal of Fisheries and Aquatic Sciences 37:130-137.

Wallace, J.B., J.W. Grubaugh, and M.T. Whiles. 1996. Biotic indices and stream ecosystem processes: results from an experimental study. Ecological Applications 61(1):140-151.

Warren, C.E. 1979. Toward classification and rationale for watershed management and stream protection. U.S.E.P.A. Corvallis, Oregon. EPA-600/3-79-059. 143 p.

Waters, T.F. 1969. Invertebrate drift--ecology and relation to stream fishes. p. 121-134. In: T.E. Northcote (ed.). Symposium on salmon and trout in streams. H.R. MacMillan lectures in fisheries. Univ. British Columbia. Vancouver, British Columbia.

Welcomme, R.L. 1979. Fisheries ecology of floodplain rivers. Longman. London. 317 p.

Welcomme, R.L. and D. Hagborg. 1977. Towards a model of a floodplain fish population and its fishery. Environmental Biology of Fishes 2(1):7-24.

Wertz, W.A. and J.F. Arnold. 1972. Land systems inventory. USDA Forest Service. Intermountain Region, Odgen, Utah.

Whiteside, B.G. and R.M. McNatt. 1972. Fish species diversity in relation to stream order and physiochemical conditions in the Plum Creek drainage basin. Am. Midl. Nat. 88(1):90-101.

Whittier, T.R., R.M. Hughes, and D.P. Larsen. 1988. The correspondence between ecoregions and spatial patterns in stream ecosystems in Oregon. Can. J. Fish. Res. Aq. Sciences 45:1264-1278.

Williams, D.D. 1990. A field study of the effects of water temperature, discharge and trout odour on the drift of stream invertebrates. Arch. Hydrobiol. 119:167-181.

Willson, M.F. and K.C. Halupka. 1994. Anadromous fish as keystone species in vertebrate communities. Conservation Biology 9(3):489-497.

Wilzbach, M.A. and K.W. Cummins. 1989. An assessment of short-term depletion of stream macroinvertebrate benthos by drift. Hydrobiologia 185:29-40.

Wolman, M.G. 1954. A method of sampling coarse river bed material. EOS Trans. AGU 35(6):951-956.

Wolman, M.G. and R. Gerson. 1978. Relative scales of time and effectiveness of climate in watershed geomorphology. Earth Surface Processes 3:189-208.

Young, M.K. 1994. Mobility of brown trout in south-central Wyoming streams. Can. J. Zoology 72(12):2078-2083

Young, M.K., W.A. Hubert, and T.A. Wesche. 1991. Selection of measures of substrate composition to estimate survival to emergence of salmonids and to detect changes in stream substrates. N. Am. J. Fish. Management 11(3):339-346.

Bibliography of Selected Literature Pertinent to Monitoring

Abt, S.R., W.P. Clary, and C.I. Thornton. 1994. Sediment deposition and entrapment in vegetated streambeds. J. Irrigation and Drainage Engineering 120(6):1098-1111.

Allen, K.R. 1971. Distinctive aspects of the ecology of stream fishes: a review. J. Fish. Res. Board of Canada 26:1429-1438.

Anderson, M.G. and T.P. Burt. 1980. Interpretation of recession flow. J. Hydrol. 46:89-101.

Armour, C.L. and W.S. Platts. 1983. Field methods and statistical analyses for monitoring small salmonid streams. FWS/OBS-83/33. Western Energy and Land Use Team, Division of Biological Services, Research and Development, Fish and Wildlife Service, US Department of the Interior, Washington, DC. 200 p.

Baker, P.F., T.P. Speed, and F.K. Ligon. 1995. Estimating the influence of temperature on the survival of chinook salmon smolts (*Oncorhynchus tshawytscha*) migrating through the Sacramento-San Joaquin River Delta of California. Can. J. Fish. Aquat. Sci. 52:855-863.

Baltz, D.M., B. Vondracek, L.R. Brown, and P.B. Moyle. 1987. Influence of temperature on microhabitat choice by fishes in a California stream. Trans. Am. Fish. Soc. 116:12-20.

Bartholow, J.M., J.L. Laake, C.B. Stalnaker, and S.C. Williamson. 1993. A salmonid population model with emphasis on habitat limitations. Rivers 4(4):265-279.

Batson, F.T., P.E. Cuplin, and W.S. Crisco. 1987. Riparian area management: the use of aerial photography to inventory and monitor riparian areas. Bureau of Land Management, Service Center, BLM/YA/PT-87/021+1737, Denver, CO. 16 p.

Beechie, T., E. Beamer, and L. Wasserman. 1994. Estimating coho salmon rearing habitat and smolt production losses in a large river basin, and implications for habitat restoration. N. Am. J. Fish. Management 14:797-811.

Beeson, C.E. and P.F. Doyle. 1995. Comparison of bank erosion at vegetated and non-vegetated channel bends. Water Resources Bull. :983-990.

Bell, J.F. and T. Atterbury (eds.). 1983. Renewable resource inventories for monitoring changes and trends. Proceedings of an international conference, August 15-19, 1983, Corvallis, Oregon, U.S.A. College of Forestry, Oregon State University, Corvallis, Oregon.

Berkman, H.E., C.F. Rabeni, and T.P. Boyle. 1986. Biomonitors of stream quality in agricultural areas: fish versus invertebrates. Environmental Management 10:413-419.

Berkman, H.E. and C.F. Rabeni. 1987. Effect of siltation on stream fish communities. Environmental Biology of Fishes 18:285-294.

Binns, N.A. and R. Remmick. 1994. Response of Benneville cutthroat trout and their habitat to drainage-wide habitat management at Huff Creek, Wyoming. N. Am. J. Fish. Management 14(4):669-680.

Bonneau, J.L., R.F. Thurow, and D.L. Scarnecchia. 1995. Capture, marking and enumeration of juvenile bull trout and cutthroat trout in small, low-conductivity streams. N. Am. J. Fish. Management 15:563-568.

Bozek, M.A. and F.J. Rahel. 1991. Assessing habitat requirements of young Colorado River cutthroat trout by use of macrohabitat and microhabitat analysis. Trans. Am. Fish. Soc. 120:571-581.

Bozek, M.A. and W.A. Hubert. 1992. Segregation of resident trout in streams as predicted by three habitat dimensions. Can. J. Zoology 70:886-890.

Brower, J.E., J.H. Zar, and C.N. von Ende. 1990. Field and laboratory methods for general ecology. Wm. C. Brown, Dubuque, Iowa. 237 p.

Buckhouse, J.C. and R.E. Gaither. 1982. Potential sediment production within vegetative communities in Oregon's Blue Mountains. J. Soil and Water Conservation (March/April):120-122.

Chen, Y.D., S.C. McCutcheon, T.C. Rasmussen, W.L. Nutter, and R.F. Carsel. 1993. Integrating water quality modeling with ecological risk assessment for nonpoint source pollution control: a conceptual framework. Wat. Sci. Tech. 28(3-5):431-440.

Chen, Y.D., S.C. McCutcheon, and R.F. Carsel. 1993. Ecological perspectives on silvicultural nonpoint source pollution control. p. 229-235. In: Proceedings of the watershed '93--A National conference on watershed management, EPA/840-R-94-002, USEPA, Washington, DC.

Chen, Y.D. and H. Chen. 1993. Determining stream temperature changes caused by harvest of riparian vegetation: an overview. p. 313-323. In: Proceedings of the conference on riparian ecosystems in the humid U.S.: functions, values and management, Atlanta, GA, March 15-18, 1993.

Chen, Y.D., S.C. McCutcheon, R.F. Carsel, A.S. Donigian, Jr., J.R. Cannell, and J.P. Craig. 1995. Validation of HSPF for the water balance simulation of the Upper Grande Ronde watershed, Oregon, USA. p. 3-13. In: G. Petts (ed.). 1995. Man's influence on freshwater ecosystems and water use. IAHS Publication No. 230.

Clary, W.P. 1995. Vegetation and soil responses to grazing simulation on riparian meadows. J. Range Management 48(1):18-25.

Cole, D.N. 1995. Experimental trampling of vegetation. I. Relationship between trampling intensity and vegetation response. J. Applied Ecology 32:203-214.

Cole, D.N. 1995. Experimental trampling of vegetation. II. Predictors of resistance and resilience. J. Applied Ecology 32:215-224.

Connell, J.H. and E. Orias. 1964. The ecological regulation of species diversity. Am. Naturalist 98:399-414.

Coon, T.G. 1987. Responses of benthic riffle fishes to variation in stream discharge and temperature. p. 77-92. In: W.J. Matthews and D.C. Heins (ed.). Community and evolutionary ecology of North American stream fishes. University of Oklahoma Press, Norman, Oklahoma.

Crisp, D.T. 1996. Measurement of stream water temperature and applications to salmonid fishes, grayling, and dace. Freshwater Biological Association. 72 p.

Cuffney, T.F., M.E. Gurtz, and M.R. Meador. 1993. Methods for collecting benthic invertebrate samples as part of the national water-quality assessment program. US Geological Survey, Open-File Report 93-406. Raleigh, North Carolina. 66 p.

Cummins, K.W., G.L. Spengler, G.M. Ward, R.M. Speaker, R.W. Ovink, D.C. Mahan, and R.L. Mattingly. 1980. Processing of confined and naturally entrained leaf litter in a woodland stream ecosystem. Limnol. Oceanogr. 25(5):952-957.

Davis, W.S. and T.P. Symon (eds.). 1995. Biological assessment and criteria. Tools for water resources planning and decision making. Lewis Publishing. [excellent, up-to-date description of bioassessment methods]

DeHart, D.A. 1975. Resistance of three freshwater fishes to fluctuating thermal environments. Unpub. M.S. thesis, Oregon State Univ., Corvallis, Oregon.

Downing J.A. 1979. Aggregation, transformation, and the design of benthos sampling programs. J. Fish. Res. Board. Can. 36:1454-1463.

Fausch, K.D. 1984. Profitable stream positions for salmonids: relating specific growth rate to net energy gain. Can. J. Zoology 62:441-451.

Gatz, A.J., Jr. 1979. Community organization in fishes as indicated by morphological features. Ecology 60:711-718.

Gebhardt, K., S. Leonard, G. Staidl, and D. Prichard. 1990. Riparian area management: riparian and wetland classification review. Bureau of Land Management, Service Center, BLM/YA/PT-91/002+1737, Denver, CO. 56 p.

Green, R.H. 1989. Power analysis and practical strategies for environmental monitoring. Environ. Res. 50:195-205.

Griffith, J.S. and R.W. Smith. 1995. Failure of submersed macrophytes to provide cover for rainbow trout throughout their first winter in the Henrys Fork of the Snake River, Idaho. N. Am. J. Fish. Management 15:42-48.

Gustafson, K.A. 1988. Approximating confidence intervals for indices of fish population size structure. N. Am. J. Fisheries Management 8:139-141.

Hayes, J.W. and I.G. Jowett. 1994. Microhabitat models of large drift-feeding brown trout in three New Zealand rivers. N. Am. J. Fish. Management 14:710-725.

Hicks, B.J., R.L. Beschta, R.D. Harr. 1991. Long-term changes in streamflow following logging in western Oregon and associated fisheries implications. Water Resources Bull. :217-226.

Jackson, D.A. and H.H. Harvey. 1989. Biogeographic associations in fish assemblages: local vs. regional processes. Ecology 70:1472-1484.

Johnson. S.L., F.J. Rahel, and W.A. Hubert. 1992. Factors influencing the size structure of brook trout populations in beaver ponds in Wyoming. N. Am. J. Fisheries Management 12:118-124.

Jones, M.L. and J.D. Stockwell. 1995. A rapid assessment procedure for the enumeration of salmonine populations in streams. N. Am. J. Fish. Management 15:551-562.

Jones, P.N. and G.A. McGillchrist. 1978. Analysis of hydrological recession curves. J. Hydrol. 36:365-374.

Keddy, P.A. 1992. Assembly and response rules: two goals for predictive community ecology. J. Vegetation Science 3:157-164.

Keleher, C.J. and F.J. Rahel. 1996. Thermal limits to salmonid distributions in the Rocky Mountain Region and potential habitat loss due to global warming: a geographic information system (GIS) approach. Trans. Am. Fish. Soc. 125(1):1-13.

Kinch, G. 1989. Riparian area management: grazing management in riparian areas. Bureau of Land Management, Service Center, BLM/YA/PT-87/021+1737, Denver, CO. 48 p.

Landres, P.B. 1995. The role of ecological monitoring in managing wilderness. Trends 32(1):10-13.
Leonard, S., G. Staidl, J. Fogg, K. Gebhardt, W. Hagenbuck, and D. Prichard. 1992. Riparian area management: procedures for ecological site inventory--with special reference to riparian-wetland sites. Bureau of Land Management, Service Center, BLM/YA/PT-92/004+1737, Denver, CO. 135 p.

Lestelle, L.C. and C.J. Cederholm. 1984. Short-term effects of organic debris removal on resident cutthroat trout. p. 131-140. In: W.R. Meehan, T.R. Merrell, Jr. and T.A. Hanley (ed.). Proceedings, fish and wildlife relationships in old-growth forests symposium. American Institute of Fisheries Research Biologists, Asheville, North Carolina.

Mahon, R. 1984. Divergent structure in fish taxocenes of north temperate streams. Can. J. Fish. Aq. Sciences 41:330-350.

Mahoney, J.M. and S.B. Rood. 1991. A model for assessing the impact of altered river flows on riparian poplars in southwestern Alberta. p. 99-104. In: S.B. Rood and J.M. Mahoney (eds.). The biology and management of southern Alberta's cottonwoods. University of Lethbridge.

Marion, D.A., S.J. Paustian, C.M. Holstine, and A. Puffer. 1987. Channel type field guide. Draft. Tongass National Forest, Chatham Area.

Meador, M.R., T.F. Cuffney, and M.E. Gurtz. 1993. Methods for sampling fish communities as part of the national water-quality assessment program. U.S. Geological Survey. Open File Report 93-104. Raleigh, North Carolina. 40 p.

Megahan, W.F., J.G. King. and K.A. Seyedbagheri. 1995. Hydrologic and erosional responses of a granitic watershed to helicopter logging and broadcast burning. Forest Science 41(4):777-795.

Menge, B.A. and A.M. Olson. 1990. Role of scale and environmental factors in regulation of community structure. Trends in Ecology and Evolution 5:52-57.

Merritt, R.W., K.W. Cummins, and R.M. Burton. 1984. The role of aquatic insects in the processing and cycling of nutrients. p. 134-163. In: Resh, V.H. and D.M. Rosenberg (eds.). The ecology of aquatic insects. Praeger Scientific, New York. 625 p.

Miranda, L.E. and W.D. Hubbard. 1994. Winter survival of age-0 largemouth bass relative to size, predators, and shelter. N. Am. J. Fish. Management 14:790-796.

Montgomery, D.R. and J.M. Buffington. 1993. Channel classification, prediction or channel response, and assessment of channel condition. TFW-SH10-93-002. State of Washington, Timber, Fish, and Wildlife, Olympia, WA. 84 p.

Moore, K.M.S., K.K. Jones, and J.M. Dambacher. 1993. Methods for stream habitat surveys: Oregon Department of Fish and Wildlife, Aquatic Inventory Project. Version 3.1, April, 1993.

Morin, A. 1985. Variability of density estimates and the optimization of sampling programs for stream benthos. Can. J. Fish. Aquat. Sci.42:1530-1534.

Myers, L.H. 1989. Riparian area management: inventory and monitoring of riparian areas. Bureau of Land Management, Service Center, BLM/YA/PT-89/022+1737, Denver, CO. 89 p.

Myers, T.J. and S. Swanson. 1995. Impact of deferred rotation grazing on stream characteristics in central Nevada: a case study. N. Am. J. Fisheries Management 15:428-439.

Overton, C.K., J.D. McIntyre, R. Armstrong, S.L. Whitwell, and K.A. Duncan. 1995. User's guide to fish habitat: descriptions that represent natural conditions in the Salmon River Basin, Idaho. Gen. Tech. Rep. INT-GTR-322. USDA Forest Service. Intermountain Research Station. Ogden, Utah. 142 p.

Parkinson, E.A., J. Barkowitz, and C.J. Bull. 1988. Sample size requirements for detecting changes in some fisheries statistics from small trout lakes. N. Am. J. Fish. Management 8:181-190. significance, power of the test

Petersen, P.H., A. Hendry, and T.P. Quinn. 1992. Assessment of cumulative effects on salmonid habitat: some suggested parameters and target conditions. TFW-F3-92-001. Center for Streamside Studies, University of Washington, Seattle, WA. 75 p.

Poff, N.L. and J.V. Ward. 1990. Physical habitat template of lotic systems: recovery in the context of historical pattern of spatiotemporal heterogeneity. Environmental Management 14(6):629-645.

Poff, N.L. and J.D. Allan. 1995. Functional organization of stream fish assemblages in relation to hydrological variability. Ecology 76(2):606-627.

Quinn, N.W.S., R.M. Korver, F.J. Hicks, B.P. Monroe, and R.R. Hawkins. 1994. An empirical model of lentic brook trout. N. Am. J. Fish. Management 14:692-709.

Rahel, F.J. 1990. The hierarchical nature of community persistence: a problem of scale. Am. Naturalist 136:328-344.

Reiman B.E. and J.D. McIntyre. 1995. Occurrence of bull trout in naturally fragmented habitat patches of varied size. Tran. Am. Fish. Soc. 124:285-296.

Ricklefs, R.E. 1987. Community diversity: relative roles of local and regional processes. Science 235:167-171.

Robichaud, P.R. and T.A. Waldrop. 1994. A comparison of source runoff and sediment yields from low- and high severety site preparation burns. Water Resource Bull. 30(1):27-34.

Shuett-Hames, D., A. Pleus, L. Bullchild, and S. Hall (eds.). 1994. Ambient monitoring program manual. Timber-Fish-Wildlife, TFW-AM9-94-001. Produced by Northwest Indian Fisheries Commission, Olympia, Washington.

Simon, A. 1995. Adjustment and recovery of unstable alluvial channels: identification and approaches for engineering management. Earth Surface Processes and Landforms 20:611-628.

Smith, B. and D. Prichard. 1992. Riparian area management: management techniques in riparian areas. Bureau of Land Management, Service Center, BLM/YA/PT-92/003+1737, Denver, CO. 44 p.

Tjomsland, T., E. Ruud, and K. Nordseth. 1970. The physiographic influence on recession runoff in small Norwegian rivers. Nord. Hydrol. 9:17-30.

Toebes, C. and D.D. Strang. 1964. On recession curves. 1. Recession equations. J. Hydrol. (N.Z.) 3(2):2-15.

Trimble, S.W. and A.C. Mendel. 1995. The cow as a geomorphic agent — a critical review. Geomorphology 13:233-253.

USFS. 1985. Fisheries habitat evaluation handbook. Monitoring. USDA Forest Service, Region 6. FSH 2609.23.

USFS. 1985. Fisheries habitat surveys handbook. Region 4. FSH 2609.23. March 1985. USDA Forest Service, Intermountain Region, Ogden, Utah.

USFS. 1990. Stream inventory handbook. Version 4.0. Region 6. Portland, Oregon.

Warren, W.G. and J.B. Dempson. 1995. Does temporal stratification improve the accuracy of mark-recapture estimates of smolt production? A case study based on the Conne River, Newfoundland. N. Am. J. Fish. Management 15:126-136.

Waters, T.F. 1969. Invertebrate drift-ecology and relation to stream fishes. p. 121-134. In: T.E. Northcote (ed.). Symposium on salmon and trout in streams. H.R. MacMillan lectures in fisheries. Univ. British Columbia. Vancouver, British Columbia.

Whiles, M.T. and J.B. Wallace. 1995. Macroinvertebrate production in a headwater stream during recovery from anthropogenic disturbance and hydrologic extremes. Can. J. Fish. Aquat. Sci. 52:2402-2422.

Williams, G.P. 1989. Sediment concentration versus water discharge during single hydrologic events in rivers. J. Hydrology 111:89-106.

Williams, G.P. and D.L. Rosgen. 1989. Measured total sediment loads (suspended loads and bedloads) for 93 United States streams. Open File Report 89-67. US Geological Survey, Denver, CO. 128 p.

Wilzbach, M.A. and K.W. Cummins. 1989. An assessment of short-term depletion of stream macroinvertebrate benthos by drift. Hydrobiologia 185:29-40.

Young, M.K. and R.N. Schmal. 1995. Mobile trout: consequences of a new paradigm. p. 185-188. In: R. Barnhart, B. Shake, and R.H. Hamre (tech. eds.). Wild trout V: wild trout in the 21st century. U.S. Government Printing Office.

Zaret, T.M. 1982. The stability/diversity controversy: a test of hypotheses. Ecology 63:721-731.

Zecharias, Y.B. and W. Brutsaert. 1988. Recession characteristics of groundwater outflow and base flow from mountainous watersheds. Water Resources Res. 24(10):1651-1658.

Zorn, T.G. and P.W. Seelbach. 1995. The relation between habitat availability and the short-term carrying capacity of a stream reach for smallmouth bass. N. Am. J. Fish. Management 15:773-783.

Appendix A

Managing Riparian Ecosystems (Zones) for Fish and Wildlife in Eastern Oregon and Eastern Washington

In March 1979 a document entitled "Managing Riparian Ecosystems (Zones) for Fish and Wildlife in Eastern Oregon and Washington" was published (Riparian Habitat Subcommittee of the Oregon/Washington Interagency Wildlife Committee 1979). The objectives of the subcommittee were to:

> *1. Describe optimum habitat conditions for fish and wildlife in riparian zones;*
>
> *2. Develop an inventory procedure to evaluate the present condition of riparian habitat, determine its potential for improvement, and provide a basis for establishment of riparian zone habitat management objectives which include fish and wildlife considerations.*
>
> *3. Recommend a policy for management of riparian zones that will ensure enhancement of fish and wildlife habitat.*
>
> *While these objectives address fish and wildlife habitat needs in riparian zones, when met they will also improve water quality, influence stream flow, and reduce stream bank erosion.* (op. cit. p. 1).

The major losses to fish and wildlife production that have been "a direct result of managing these zones without established fish and wildlife habitat objectives" (op.cit. p. 1) were the impetus for adopting the standards proposed in this agency consensus document. Because 80% of terrestrial wildlife species in northeastern Oregon are "either directly dependent on riparian habitat or utilize it proportionately more than any other habitat type (Thomas et al. 1977)" (op. cit. p. 2), riparian protection should be key to total ecosystem restoration as well as listed fish restoration. Despite the fact that "the importance and relationships of riparian vegetation to fish and wildlife habitat, water quality and quantity, and erosion control have been known for years" (op. cit. p. 2), forest management still remains largely devoid of firm standards or holistic management and is mostly a matter of planned removal of all but "key" components or logging justified on the basis of forest health or improvement in fish and wildlife values. Many forest-wide standards that are on the books are essentially meaningless because there has never been any intention of monitoring them (e.g., riparian soil compaction).

This subcommittee identified streambed sedimentation as a primary determinant of fish production potential because of the importance of clean gravel/rubble as spawning habitat and a source of food production. Even in 1979 these agency personnel representing the USFS, BLM, SCS, USFWS, ODFW, and WDG noted that "trout populations can be further reduced if pools become filled with sediments thus eliminating rearing or hiding habitat" (op. cit. p. 5). They adopted a standard of no more than 15% inorganic sediment <3.3 mm, very similar to the standard proposed in Rhodes et al. (1994). Their graph indicating the relationship between percentage potential trout production and fine sediment indicates that production is >90% of potential when fine sediment is <15% fines and drops to 70% at 20% fines, and 20% of potential at 40% fines. "While the data were developed for trout, they apply equally to salmon and steelhead rearing in fresh water" (op. cit. p. 2).

R.E. Worthington, USFS Regional Forester indicated that the USFS would adopt these procedures in their planning process and that they were "consistent with Forest Service direction." He stated that "[b]y describing the present on-site habitat conditions and projecting the potential of the site, the trade-offs involved with other resources can be addressed and management prescriptions written based on recommended riparian habitat conditions that are responsive to overall Forest Plan objectives."

Despite the fact that recognition of the control of fine sediment on fish production remains the same today as it was in 1979 and the USFS and other agencies saw it as an important management tool consistent with their direction, it has somehow been disassociated from their direction today. This is possibly the result of identifying the tradeoffs that would be involved if aquatic resources were actually managed for high levels of both production and habitat quality and of determining costs of fully protecting beneficial uses. This is, unfortunately, the principal manner in which adaptive management is practiced by federal land managers today: that is, on the basis of experience in observing linkages between management actions and impacts in stream channels, decisions are made to ignore those habitat parameters that present challenges in monitoring or produce monitoring results that would limit land management activities. In the absence of monitoring data on key habitat parameters, blind faith in BMPs seems to be required. Also, blind faith is required in order to believe predictions of very slow, long-term improving trends that could occur while continuing land disturbing activities that are only slightly reduced in intensity from the levels that led to severe habitat degradation. Abandonment of this landmark agreement, which is supported by decades of technical literature, followed by the current emphasis on "holistic" or ecosystem management, appears hypocritical when one realizes that the most significant source of habitat degradation in the Snake basin is not even monitored.

Appendix B

Effects of Land Use on Salmon Habitat

Summary of effects of land use activities on key portions of the watershed system (in-stream, streambank, riparian, hillslope, watershed) and their linked effects on specific aspects of salmon habitat. Although land use activities can have numerous other effects on aquatic resources, primary attention was given to effects on habitat variables for which standards are proposed. Key: "↑" = increase; "↓" = decrease; "⇒" = "a function of;" CE = cobble embeddedness; LWD = large woody debris. (Table reproduced from Rhodes et al. 1994).

Land use activity	System component affected	Mechanism of effect	Habitat parameter affected
grazing	streambank	↑ bank erosion ⇒ ↓ vegetation density and rate of regrowth; ↑ sediment delivery	↑ CE, ↑ surface fines, ↓ streambank stability
		↑ bank trampling and calving; ↓ bank stability	↓ bank overhang
		↑ summer water temperature, ↓ winter water temperature, ↑ daily and seasonal extremes, ↑ headward movement of critical thermal maxima	↑ thermal loading (summer) interception, ↑ heat loss in winter; ⇒ ↑ channel width ⇒ ↑ sediment load, ↑ bank trampling and ↓ bank stability;
		↓ wet meadow vegetation ⇒ grazing, soil compaction;	↑ solar load interception
		↓ shrubs and trees, shift in vegetative community; ⇒ ↓ shrub canopy, ↑ trampling and cropping of seedlings/saplings ⇒ ↑ livestock grazing pressure	↓ stream shading
		↑ bank erosion, ↓ bank stability ⇒ ↓ vegetation density and ↓ rate of regrowth; ↑ sediment delivery; ↑ physical disturbance of soil surface	↓ pool volume

Table 2 (Continued). Summary of effects of land use activities on aspects of salmon habitat.

Land use activity	System component affected	Mechanism of effect	Habitat parameter affected
grazing	floodplain	↑ solar radiation interception, ↑ longwave radiation emission from water surface ⇒ riparian vegetation cropping and prevention of regrowth	↑ summer water temperature, ↓ winter water temperature, ↑ daily and seasonal extremes, ↑ headward movement of critical thermal maxima
		↓ groundwater baseflow ⇒ ↑ channel incision, ↑ soil compaction, ↓ infiltration capacity, ↑ overland flow, ↓ water storage capacity, wetland damage	↑ maximum water temperature, ↑ water temperature extremes, ↓ flow velocity, ↑ intermittent reaches
		↓ wetland area ⇒ ↑ streambed incision, ↑ baselevel lowering of wetland, ↑ drainage of wetland	↑ maximum water temperature, ↓ bank stability
		↑ upstream channel erosion, ↑ sediment load ⇒ all streambank, riparian, and floodplain grazing effects	↑ CE and ↑ surface fines, ↓ pool volume and frequency
	hillslope	↑ peakflow ⇒ ↑ vegetation removal by grazing, ↑ soil compaction	↑ sediment delivery, ↓ bank stability
logging	riparian	↓ stream surface shade, ↑ solar input, ↑ nighttime longwave radiation flux; ⇒ ↓ vegetation cover	↑ summer water temperature, ↓ winter water temperature, ↑ daily and seasonal extremes, ↑ headward movement of critical thermal maxima
		↑ channel widening, ↑ streambank erosion; ⇒ ↓ vegetation cover, ↑ peakflow, ↓ abundance of deep rooted vegetation, ↑ surface and mass erosion	↑ CE and ↑ surface fines, ↓ pool volume and frequency, ↓ streambank stability

Table 2 (Continued). Summary of effects of land use activities on aspects of salmon habitat.

Land use activity	System component affected	Mechanism of effect	Habitat parameter affected
		↑ channel and bank erosion, ↑ sediment delivery ⇒ ↑ vegetation removal, ↓ LWD volume and recruitment to channel (↓ sediment storage)	
logging	riparian	↓ LWD volume and recruitment rate ⇒ ↑ vegetation removal, ↓ size and number of trees	↓ pool volume and frequency, ↓ residual pool depth
		↑ pool sedimentation ⇒ ↑ magnitude and frequency of channel erosion, ↓ LWD	
		↓ pool volume ⇒ ↑ pool sedimentation and ↓ LWD; ↑ channel width ⇒ ↑ discharge peaks, ↑ sediment delivery, ↓ streambank stability	↑ summer water temperature, ↓ winter water temperature, ↑ daily and seasonal extremes, ↑ headward movement of critical thermal maxima
		↑ vegetation cover, basal area, and deep rooting density, ↑ soil disturbance	↑ sediment delivery, ↑ turbidity
		↓ size and number of trees; ↓ LWD recruitment ⇒ long recovery time of riparian trees	↓ LWD recruitment
	hillslope	↑ erosion and sediment delivery; ↑ mass failures ⇒ ↓ rooting strength, ↓ tree cover	↑ CE and ↑ surface fines, ↓ pool volume and frequency
		↓ vegetation cover, ↓ groundcover, ↓ root strength, ↑ soil disturbance, ↑ mass and surface erosion, ↑ peakflow, ↑ overland flow	↑ sediment delivery, ↑ turbidity
		↑ peakflow ⇒ ↑ vegetation removal, earlier snowmelt	↑ sediment delivery, ↓ bank stability

Table 2 (Continued). Summary of effects of land use activities on aspects of salmon habitat.

Land use activity	System component affected	Mechanism of effect	Habitat parameter affected
	watershed	↑ peak flows	↑ sediment delivery
		logging moratorium	↓ sediment delivery
roads	riparian	↓ stream surface shade, ↑ solar input, ↑ nighttime longwave radiation flux; ⇒ ↓ vegetation cover	↑ summer water temperature, ↓ winter water temperature, ↑ daily and seasonal extremes, ↑ headward movement of critical thermal maxima
		↑ channel widening, ↑ streambank erosion; ⇒ ↓ vegetation cover, ↑ peakflow, ↓ abundance of deep rooted vegetation, ↑ surface and mass erosion originating from road corridor	↑ CE and ↑ surface fines, ↓ pool volume and frequency, ↓ streambank stability
		↑ channel and bank erosion, ↑ sediment delivery ⇒ ↑ vegetation removal, ↓ LWD volume and recruitment to channel (↓ sediment storage)	
		↓ LWD volume and recruitment rate ⇒ ↑ vegetation removal, ↓ volume recruitment of LWD of large length and diameter	↓ pool volume and frequency, ↓ residual pool depth
		↑ pool sedimentation ⇒ ↑ magnitude and frequency of channel erosion, ↓ LWD	
		↓ pool volume ⇒ ↑ pool sedimentation and ↓ LWD; ↑ channel width ⇒ ↑ discharge peaks, ↑ sediment delivery, ↓ streambank stability	↑ summer water temperature, ↓ winter water temperature, ↑ daily and seasonal extremes, ↑ headward movement of critical thermal maxima
		↑ vegetation cover, basal area, and deep rooting density, ↑ soil disturbance	↑ sediment delivery, ↑ turbidity
		↓ size and number of trees; ↓ LWD	↓ LWD recruitment

Table 2 (Continued). Summary of effects of land use activities on aspects of salmon habitat.

Land use activity	System component affected	Mechanism of effect	Habitat parameter affected
		recruitment ⇒ long recovery time of riparian trees	
roads	hillslope	↑ erosion and sediment delivery from road surfaces; ↑ mass failures originating from road system	↑ CE and ↑ surface fines, ↓ pool volume and frequency
		↓ vegetation cover, ↓ groundcover, ↓ root strength, ↑ soil disturbance, ↑ mass and surface erosion, ↑ peakflow, ↑ overland flow↑ sediment delivery, ↑ turbidity	
		↑ peakflow ⇒ ↑ vegetation removal, earlier snowmelt, ↑ water routing from road drainage system	↑ sediment delivery, ↓ bank stability
	watershed	↑ road density, ↑ peak flows	↑ sediment delivery
		logging moratorium, ↓ road density	↓ sediment delivery
mining	riparian	↑ erosion and transport of mine tailings, ↓ size and number of trees, ↓ LWD recruitment ⇒ long recovery time of riparian trees	↑ CE and ↑ surface fines, ↓ pool volume and frequency
	hillslope	acid mine runoff, heavy metal pollution, sediment delivery	↓ water quality
	in-stream	gravel extraction	↑ CE and ↑ surface fines, ↓ pool volume and frequency, ↑ turbidity

Table 2 (Continued). Summary of effects of land use activities on aspects of salmon habitat.

Land use activity	System component affected	Mechanism of effect	Habitat parameter affected
agriculture	riparian	intense soil disturbance, ↓ vegetation cover, ↓ vegetation rooting density	↑ CE and ↑ surface fines, ↓ pool volume and frequency, ↑ sediment delivery
		feedlot runoff, runoff of agricultural chemicals (fertilizers, pesticides)	↓ water quality
	watershed	↓ stream summer baseflow ⇒ irrigation withdrawal or diversion	↑ summer water temperature
		intense soil disturbance, ↓ vegetation density	↓ pool volume and frequency, ↑ sediment delivery

Appendix C

Coarse Screening Process Summary of Standards

(table reproduced from Rhodes et al. 1994)

In-Channel Habitat Conditions

Element and Standard	Condition	Related Land Management Standards
Channel Substrate		
Surface fine sediment (diam.<0.25 in.): Average surface fine sediment ≤20% in spawning habitat; no increase where already ≤20%.	Substrate standards met and estimated sediment delivery ≤20% over natural.	No increase in sediment delivery from single or combined activities. All new activities with potential to produce sediment offset by at least equivalent sediment abatement via active restoration.
Cobble embeddedness (CE): Average CE ≤30% in rearing habitat; no increase where already ≤30%.	Substrate standards met but estimated sediment delivery >20% over natural	Reduce sediment delivery via passive and active restoration[1] to ≤20% over natural. All new activities that increase erosion should be combined with sediment abatement measures that result in a net reduction in sediment delivery.
	Substrate standard exceeded and estimated sediment delivery >20% over natural; fine sediment or CE increase.	Reduce sediment delivery via complete passive restoration until sediment delivery ≤20% over natural and substrate conditions meet standards or exhibit a statistically significant (p<0.05) improving trend over ≥5-yr period. Implement active restoration as needed to reduce sediment delivery and improve substrate conditions.
	Substrate standard exceeded and estimated sediment delivery ≤20% over natural.	Reduce sediment delivery via complete passive restoration until substrate conditions meet standards or exhibit a statistically significant (p<0.05) improving trend over ≥5-yr period. Implement active restoration as needed to reduce sediment delivery and improve substrate conditions.
Fines by depth: Although not set as a numeric standard, monitoring substrate trends at depth is recommended as part of adaptive management.	Regardless of condition.	Monitor fines by depth in key areas to evaluate correlation with surface fines by area.
Channel Morphology		

Coarse Screening Process Summary of Standards (Continued)

In-Channel Habitat Standards

Element and Standard	Condition	Related Land Management Standards
Large Woody Debris (LWD): Land management standards set in lieu of numeric standard.	Regardless of condition.	Fully protect vegetation and soils within riparian reserves, meet bank stability standards, and Monitor LWD. Implement active restoration as needed to reestablish natural LWD recruitment from riparian zones. Add LWD to streams only after causes of LWD loss have been adequately addressed and where ecologically appropriate.
Pool frequency and volume: Land management standards set in lieu of numeric standard.	Regardless of condition.	Fully protect vegetation and soils within riparian reserves, meet bank stability standards and limit/reduce sediment delivery to ≤20% over natural. Monitor pool frequency and volume. Add LWD only after causes of pool loss have been adequately addressed and where ecologically appropriate.
Residual pool volume: Not set as a numeric standard. Achieve an increasing trend in residual pool volume.	Regardless of condition.	Fully protect vegetation and soils in riparian reserves, meet bank stability standards, and limit/reduce sediment delivery to ≤20% over natural. Monitor residual pool volume.
	Declining trend from baseline.	Reduce sediment delivery via passive and/or active restoration.
Bank stability: Bank stability on all streams average ≥90%; no decrease in bank stability when >90%.	Average bank stability <90% or a decrease in bank stability.	Implement passive restoration (suspension of grazing within half a tree height of floodplains or streams) until bank stability meets standard or exhibits a statistically significant ($p<0.05$) improving trend over ≥5 years. Implement active restoration addressing causes of bank instability as needed to improve bank stability. Do not mechanically stabilize banks (e.g., rip-rap).
	Average bank stability ≥90%.	Apply appropriate management controls to maintain bank stability.

In-Channel Habitat Standards

Element and Standard	Condition	Related Land Management Standards
Water Quality		
Water temperature: Maximum daily water temperature ≤60°F in historically usable spawning and rearing habitat.	Maximum daily water temperature >60°F.	Fully protect vegetation in riparian reserves. Implement complete passive riparian restoration to reduce water temperature. Suspend grazing within riparian reserves until water temperature meets standard or exhibits a statistically significant improving trend over ≥5 years. Implement active restoration as needed to improve water temperature conditions.
	Maximum daily water temperature ≤60°F.	Fully protect vegetation within riparian reserves from any additional impacts. Control and monitor on-going activities within riparian reserves to assure they do not increase water temperatures.
Stream shading: Land management standards set in lieu of a numeric standard.	Regardless of condition.	Fully protect vegetation within riparian reserves. Activities that decrease shading or forestall recovery should not be allowed on any stream.
Misc. Pollutants: Review and revise current state and federal water quality standards as needed to adequately protect salmon. In the interim, meet current water quality standards.	Regardless of condition.	Monitor water quality parameters set as state and federal water quality standards where the potential for pollution exists. Eliminate or restrict transport of toxic materials along and upstream of spawning and rearing reaches. Eliminate storage of toxic materials within watersheds with spawning or rearing habitat.
Water Quantity and Timing: Not set as numeric standards.	Regardless of condition.	Suspend additional groundwater and surface water withdrawals in watersheds with spawning and rearing habitat until studies determine instream flows are more than adequate for salmon production and survival, and maintenance and restoration of favorable habitat conditions. Where flows are inadequate, acquire instream flows. Protect and restore wetlands and degraded meadow systems. Fully protect vegetation and soils within riparian and roadless reserves.

Coarse Screening Process Summary of Standards (Continued)

Land Management Standards

Element and Standard	Condition	Related Land Management Standards
Sediment delivery: Sediment delivery ≤20% over natural.	(Additional details under **Channel Substrate**)	
Riparian reserves: ≥300 ft slope distance from floodplains, or stream where floodplains do not exist, or to topographic divide, whichever is less.	Regardless of condition.	Fully protect vegetation and soils within riparian reserves from any additional anthropogenic disturbance. Do not implement approaches to riparian restoration involving vegetation removal until they have been shown to be effective under applicable ecological conditions. Reduce existing road mileage in reserves. Improve road drainage within reserves. Active restoration should focus on reducing impacts within riparian reserves where needed to improve habitat conditions.
	Water temperature standard exceeded.	Suspend grazing within riparian reserves until water temperatures meet standard or exhibit a statistically significant improving trend over at least 5 years.
	Bank stability standard not met.	Suspend grazing within half a tree height from the edge of floodplains, or streams when floodplains are absent, until bank stability meets standard or exhibits a statistically significant improving trend over at least 5 years.
	All habitat standards met.	Carefully control and monitor all on-going activities within riparian reserves to assure degradation does not occur.

Coarse Screening Process Summary of Standards (Continued)

Land Management Standards

Element and Standard	Condition	Related Land Management Standards
Equivalent Clearcut Area (ECA): Not recommended as a land use standard for limiting land disturbance.	Limit land disturbance via riparian and roadless reserves, application of in-channel habitat standards, and sediment delivery standard.	
Roads: Decrease road mileage in managed watersheds. Improve drainage and decrease sediment delivery from roads that will not be obliterated or re-located.	Habitat conditions in <90% of managed watersheds either meet standards or have exhibited statistically significant improvement.	Defer construction of new roads. Continue to upgrade, obliterate, or re-locate existing roads.
	Habitat conditions in >90% either meet all habitat standards or have exhibited statistically significant improvement.	Consider re-evaluation of prohibition on new road construction.
Grazing: Forage utilization standards not recommended as a numeric standard.	Regardless of condition	Eliminate livestock access to spawning reaches during spawning and incubation periods.
	Substrate standards exceeded.	Suspend grazing within watershed until sediment delivery is reduced via passive and active restoration to $\leq$20% over natural and substrate conditions meet standards or exhibit a statistically significant improving trend over $\geq$5-yr period.
	Bank stability standard not met.	Suspend grazing within half a tree height from the edge of floodplains, or streams when floodplains are absent, until bank stability meets standard or exhibits a statistically significant improving trend over at least 5 years.
	Water temperature standard exceeded.	Suspend grazing within riparian reserves until water temperatures meet standard or exhibit a statistically significant improving trend over at least 5 years.

Coarse Screening Process Summary of Standards (Continued)

Land Management Standards

Element and Standard	Condition	Related Land Management Standards
	All habitat standards met.	Carefully control and monitor all grazing within riparian reserves to assure degradation does not occur.
Roadless reserves: Maintain all roadless tracts >1000 acres undisturbed; maintain smaller roadless tracts undisturbed until documented that disturbance will not forestall habitat recovery nor foreclose management options.	Habitat conditions in <90% of managed watersheds either meet all habitat standards or have exhibited statistically significant improvement.	Maintain protection of roadless tracts.
	Habitat conditions in >90% of managed watersheds either meet all habitat standards or have exhibited statistically significant improvement.	Consider re-evaluation of roadless reserves.
Spatial Criteria for Re-evaluation of Land Management Standards: Habitat conditions in >90% of managed watersheds either meet all habitat standards or have exhibited statistically significant improvement.	Habitat conditions in <90% of watersheds meet all habitat standards or have exhibited a statistically significant improving trend in all habitat elements over $\geq$5-yr period.	Maintain integrity of riparian and roadless reserves and continue to implement screening process. Monitor trends in habitat condition for the entire set of watersheds within the Snake River Basin for trends. Implement active restoration as needed to improve habitat conditions.
	Habitat conditions in >90% of watersheds meet all habitat standards or have exhibited a statistically significant improving trend in all	Consider re-evaluation of land use standards (e.g., riparian and roadless reserves).

habitat elements over $\geq$5-yr period.

Endnotes for Summary

1. Passive restoration is defined as curtailing or deferring activities that contribute to degraded conditions or forestall natural recovery processes. <u>Complete</u> passive restoration is defined as curtailing and deferring all activities at the watershed scale that contribute to degradation or potentially forestall recovery. In the case of channel substrate, complete passive restoration entails curtailing and deferring all ground disturbing activities that increase erosion over natural levels.

Active restoration is defined as taking actions to reduce or eliminate the effects of existing impacts, such as obliterating or upgrading roads.

Appendix D

Coarse Screening Process Decision-Making Process

Flow chart of decision-making process for application of the Coarse Screening Process to land management activities in watersheds supporting ESA-listed fish stocks (Rhodes et al. 1994, as amended by R. Beaty, CRITFC).

HIGH-RISK ACTIVITIES [a]

MODERATE-RISK ACTIVITIES [b]

Are sufficient data available on in-channel habitat and land management in **all managed watersheds** within the spawning and rearing habitat of the listed species?

No →

Defer or curtail all activities that can potentially affect attainment of standards until further baseline data are collected and summarized.

← No

In this watershed, are sufficient data available for **in-channel habitat** variables used as standards?

Yes ⇓

← No

Are sufficient data available for **condition/land management** of this watershed?

Yes ⇓ (high-risk)

Yes ⇓ (moderate-risk)

≥90% of all managed watersheds within the spawning and rearing habitat of the listed species meet all standards for in-channel habitat and land management **or** have a documented , statistically significant improving trend in all in-channel habit variables used as standards?

No →

Defer or curtail all activities that can forestall recovery. Continue efforts to meet all habitat standards; undertake active restoration activities as needed. Monitor habitat parameters to document changes in conditions.

← No

Do in-channel habitat conditions in this watershed meet all standards or show a significantly significant improving trend in all variables used as standards?

Yes ⇓

← No

Does land management in this watershed meet or exceed all screening standards?

Yes ⇓ (high-risk)

Yes ⇓

Proceed with regulatory review.

← No

Will the proposed activity violate screening standards?

Yes ⇓

Defer or modify proposed activity.

← No

Can the screening standards be relaxed yet still maintain habitat conditions conducive to excellent salmon survival?

Yes ⇓

Consider relaxing specific land management standard(s) in this watershed to accommodate proposed activity.

a High-risk activities include road building, activities in roadless reserves, logging of riparian reserves, livestock access to spawning areas, etc. See Rhodes et al. 1994, Table B, or p. 1 of this document.

b Moderate-risk activities include upland grazing when sediment stds are met, upland logging when sediment and temperature stds are met, etc. See Rhodes et al. (1994), Table B, or p. 1 of this document.

Appendix E

Need for Holistic Monitoring Plans: A Critical Failure of Today's Federal Plans

Despite the recognition of the important role of excessive fine sediment in degrading habitat quality in spawning and rearing areas (USFS 1994, 1995), fine sediment has not been identified as a necessary monitoring parameter (see USFS 1994, 1995, **PACFISH**), even in the Idaho batholith where inundation of spawning gravels by management-generated sediment erosion is legendary. The monitoring plan devised by the National Aquatic Monitoring Center (USFS 1994, 1995) ignores measures of channel bed sedimentation (percent fines, embeddedness) and instead believes that width/depth, bank stability, and lower bank angle (incidence of bank overhang) will suffice as warning devices. The incidence of bank overhang in a reach is an indication of the severity of livestock damage to stream channels. Sediment generated from this action can be deposited in the adjacent reach. However, streambed sedimentation is not solely a reach phenomenon; it reflects the cumulative effects of all natural events and management activities upstream on inputs and to a limited extent the backwater effects from downstream damming (LWD, artificial dams). Consequently, it is unlikely that such surrogates for increase in bed fine sediment will be adequate. Changes in W/D and loss of pools are expected under increasing sediment loads to channels, but these effects, though biologically significant, are generally slow, progressive changes relative to the changes occurring in surficial bed sediments. Likewise, during improving trends in habitat quality, surface fines are rapid to respond to sediment load reductions, whereas fines at depth are slower. Slower still to recover are pool volumes and W/D. The biological significance of protecting bank overhangs, pool volumes, and narrow channels and the relatively slow response times of these elements imply that it is critical to prevent channel damage rather than attempting to effect restoration. This involves elimination of livestock trampling and riparian disturbance from logging and roading of these areas.

It is clear that elevated levels of fine sediment and embeddedness over natural in spawning and rearing areas have increasingly negative biological consequences of their own. To expect that surrogate measures having significantly different response times, other physical stressors leading to their responses, and operating at predominantly the reach scale can provide the rapid, direct, early warning benefit of measuring channel sediment composition is a gross oversight.

Few instream habitat quality measures other than water temperature have been so widely studied as effect of fine sediments. Chapman and McLeod's (1987) review of fine sediment effects on salmonids found a universally negative trend in survival to emergence with increasing levels of fine sediment. This general trend was similar when comparing both laboratory and field studies. In addition, Huntington (1994), after surveying 1320 distinct stream reaches in the Clearwater National Forest, Idaho between 1989 and 1993 comprising 1090 km in total length, reported that the average substrate composition for managed C-channels had 57.8% fine sediment, while that in unroaded C-channels had 24.1% fine sediment. In B-channels the managed vs. unroaded comparison for percentage fine sediment is about 5% to 24%. Huntington pointed out that even the unroaded streams may not be behaving at the low end of their inherent fine sediment range because of lingering effects of historic wildfire. This work underscores that low gradient channels suffer the highest severity biological impact of cumulative sediment deposition, but it is important for monitoring programs to recognize early warning signals of sediment accumulation in higher gradient channels to prevent downstream effects. Although C-channels are considered to be prime spawning and rearing areas for chinook, Huntington found that chinook rearing densities were 25x greater in unroaded C-channels than in managed ones; B-channels were different by a factor of >10x. In addition to percentage fine sediment differences between unroaded and managed channels, embeddedness in unroaded C-channels averaged 49.1% but was 72.4% in managed C-channels. B-channels had corresponding values of 26 and 49%, respectively.

Despite the overwhelming evidence as to the negative influence of increasing fine sediment levels beyond natural levels, detractors cite scientific uncertainties present in every aspect of fishery sciences as reasons (excuses) for ignoring the method. The menu of excuses can be mixed and matched in numerous combinations so that one piece of flawed logic supports the next. According to the menu selected, one can justify adopting standards so extreme that violations will seldom be observed, discounting biological consequences, or measuring other variables that are easier and are purported to be somewhat correlated with the variable needing measurement.

> <u>A menu of excuses and scientific red herrings.</u> Is there really a plateau in the response curve between 0-20% fines within which negative effects are minor? If so, this should mean that 20% fines is just as good as 15%.
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> Can't we really increase fines to maybe 22 or 25% with little marginal increase in mortality?
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> If fines are naturally high in a pristine stream, can't we assume that fish are adapted to whatever conditions are present and that we can always allow sediment to increase by at least 10% with no consequences.
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> If we attempt to play environmental brinkmanship (so that timber and range commodities can be optimized), can't we set the standard to the maximum level (20%) as in most Idaho Forest Plans (or better yet set it at 30% outright as with the Challis N.F.) but not actually measure sediment (PACFISH), not use any sediment modelling procedures for estimating cumulative watershed sediment yields but rely on professional judgement, implement a monitoring program that detects W/D changes downstream in C-channels (even though lag times are very long) as a means of employing adaptive management, and if W/D deviates beyond confidence limits for all regional reference streams (most of which are already heavily altered and include outliers that expand the performance range), try to trace the source upstream and see whether something should be modified (assuming funds are available to monitor the C-channel).
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> If we have desired levels of LWD, won't this eliminate the need to control sediment because fines will be stored behind LWD and local zones of sediment sorting and routing will create patches of suitable spawning gravel, at least for some fish.

Measuring sediment is desirable but it is too hard. It requires numerous samples, expensive equipment, laborious collection and sample processing. Sediment conditions are spatially and temporally variable, making results highly variable and difficult to assess statistically significant changes from one time to the next. Many technicians do a poor job collecting sediment data and there is variation among observers in ocular estimates of sediment making the data useless for comparative purposes.

Maybe we should skip measuring sediment and measure inter-gravel dissolved oxygen instead because lack of oxygen is what really kills incubating eggs, oxygen is so much easier to measure, and avoids spatial/temporal sampling problems.

It is unrealistic to expect that all streams would be harmed by adding extra sediment. Some streams are sediment poor, some sediment rich. The poor ones obviously need more. With the rich ones, a little more would be imperceptible visually or statistically. Debris torrents have been shown to provide pools where they come to rest in mainstem channels. These are often "hot spots" biologically because they provide habitat for fish in streams where LWD has been removed, causing a loss of spawning gravels too. For this reason, debris torrents and sediment entry to streams are good for fish. Because it is scientifically unjustified to make broad brush generalizations about the harm from sediment, we must conclude that the sediment issue is overblown.

The world is a very complex place. There are so many species, races, stocks, life stages, each with different adaptations to environment. How can we generalize from lab studies to the field. Maybe each local population is uniquely adapted to the conditions it is found in. If we find that 20% fine sediment is harmful in one field situation, it would be scientifically unsound to state that these conditions are apt to be harmful in another location, even for the same species.

The burden of proof should be on the fish and their supporters to show beyond a shadow of a doubt (e.g., 95% or 99% confidence limits) that negative changes are occurring in fine sediment over a several year period before existing management actions are modified in any way. It is unfair that improving trends be demonstrated using the same statistical criteria before current management actions (e.g., logging, grazing, etc.) are allowed to continue because we can be totally certain that today's BMPs will result in some degree of recovery given enough time. We know the degree of effectiveness of BMPs under all probable environmental conditions and combinations of management actions so we are uniquely able as managers to balance all resource outputs. Fish may be present in small numbers but that is not primarily the fault of habitat. Other factors such as the dams are more important. These small populations of fish can hold on for decades while the BMPs are applied and habitat improves on its slowly ascending glide path.

> It doesn't really matter if fine sediment levels reach even 40% fines because salmon will clean the gravels in redd construction. If we had a lot of spawners they could effectively clean an embedded reach. After spawning, we can assume that the redds will remain clean throughout the winter incubation period even if ambient sediment levels are high. If sedimentation occurs on the redd surface, infiltration will only take place in the top few centimeters and sediment bridging will not allow penetration all the way to the egg pocket. Even though the redd may be totally covered in fine sediment, inter-gravel dissolved oxygen and water flow may still remain high. Even though bedload movement is high in a stream system with elevated sediment delivery, scouring will not occur to egg pocket depth and even if scouring occurs, the spatial variability of scour is large so not all redds will be affected.

Excuses for not measuring or modeling sediment deposition and yields are as abundant as are excuses for not applying our best judgement about what we know and extrapolating. Every one of the adages just mentioned has so many conceptual flaws that a document could be written simply peeling away the myths about sediment monitoring. In fact, an exposition to many of these claims does exist in the CSP document.

For the most part, the same kinds of technical concerns leveled against sediment monitoring exist with regard to other monitoring parameters. This is not to say that no technical difficulties exist in sediment monitoring or in any other monitoring issue. But ignoring the issue or finding other more interesting parameters to measure that can be called surrogates does not further the ability to manage land and aquatic resources. Every effort must be made to ensure that monitoring programs are comprehensive in their representation of critical stream physical processes, spatially extensive and methodologically intensive in order to fully document coarse scale condition and adequately represent fine scale processes on sensitive or important stream or watershed components, and employ variables that are attuned to short- as well as long-term system responses to provide early warning stress indicators and long-term recovery indices.