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A Review of Strategies for Recovering Tributary Habitat

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Background

Whether tributary habitat improvements have achieved, or are likely to achieve, the goal of recovering conditions favoring the natural production of native salmonids in the Columbia River Basin is an open question, and a review of the approaches, assessment procedures, and implementation strategies for habitat improvement is therefore timely. In this report we examine several topics central to the recovery of tributary habitat: (1) the biological objectives related to habitat recovery, (2) the strategies for implementing restoration; (3) the incentives for implementing restoration; (4) the scientific foundation for habitat recovery; and (5) monitoring and evaluation. The objective of the review is to answer the question: *What concepts and strategies should be incorporated in habitat recovery actions to improve their chances for success?*

Biological Objectives

Biological objectives are important because they provide measurable targets for habitat recovery. Tributary performance standards are referenced in the All-H Report and are taken to mean the habitat conditions that would achieve biological recovery in an area of interest. These standards become the *de facto* biological objectives for tributary habitat. Fixed habitat standards (the traditional approach to habitat protection) do not easily account for natural environmental variation, differences in habitat requirements among species, or changes in habitat needs over a fishes' life cycle. Additionally, managers often treat fixed standards as targets, tolerating habitat degradation until a danger threshold is exceeded and only then taking corrective action.

Habitat standards based on the distribution of conditions observed in unmanaged watersheds are more ecologically relevant, especially if they are expressed at appropriately large scales of space and time. Large spatial and temporal scales are required to account for the variety of conditions generated by natural disturbance and recovery. In many cases, decades or centuries might be required for habitat attributes within a degraded watershed to achieve a distribution of conditions comparable to those occurring in an unmanaged watershed. Therefore, performance standards are usefully articulated by comparing the distribution (median, range, and variance) of current habitat conditions to the desired distribution of habitat conditions (based on studies of reference watersheds), and tracking the rate at which conditions in a restored area converge on those observed in unmanaged landscapes. Standards expressed in this manner have the advantage of providing meaningful feedback on the efficacy of restoration much more realistically than a simple comparison of current conditions to a fixed set of habitat standards. While we acknowledge that the distribution of natural habitat conditions is often imperfectly known, and therefore do not advocate complete abandonment of habitat and water quality standards, we argue that a better understanding of natural variability provides a much more sound context for defining biological objectives for tributary habitat.

Implementation Strategies

With accurate data on habitat condition and sensitivity of fish populations to human actions, numerous analytical methods can be used to develop habitat recovery plans. Significant data gaps exist, however, and very few analytical methods have been sufficiently verified by field studies. As a consequence, several predictive methods have been developed to use incomplete data to relate tributary habitat to aquatic community condition or the abundance of target species. These methods are of three basic types: expert opinion, expert systems, and empirical models. Expert opinion has been most commonly applied in the Columbia Basin to habitat restoration projects. This approach is highly subjective and the assumptions underlying the opinions are often not field verified, or even made explicit. As a result, the products of such an approach are generally less helpful than more formal decision processes, especially those using quantitative information. Expert systems such as Ecosystem Diagnosis and Treatment (EDT) represent an improvement over undisciplined application of expert opinion. They provide a mechanism for objectively, and in some cases transparently, combining opinions from multiple scientists. They can also provide a clear indication of the assumptions underlying model predictions, if the process is properly documented. Parameterizing these models can provide a very good indication of significant data gaps, thus highlighting areas where more data are needed.

However, model outputs must be interpreted with a full appreciation of the limitations imposed by the model structure and underlying assumptions; in most cases they have not been tested in the area to which they have been applied. Empirical models relate the abundance or occurrence of the species of interest to habitat attributes at a given location, and in this sense are directly applicable to restoration decisions because they are based on local data. In empirical models, associations between populations and habitat characteristics can be used to predict the distribution and abundance of a species or a species assemblage for locations where only habitat information is available. The predicted distribution directs protection and restoration actions to locations with the greatest potential (i.e., the appropriate suite of habitat conditions) to support target species. Empirical modeling techniques are informative when relatively complete data for fish populations and habitat are available, but they can be used even when data are incomplete. The ability to use empirical models has been enhanced by new statistical approaches and a steady improvement in remote sensing and spatial mapping technologies.

Until the accuracy of various decision support approaches is properly evaluated on the ground, the ISAB believes the best approach to the development of habitat recovery plans is to use more than one analytical method. Policy makers can gauge the degree of scientific uncertainty by comparing the output from two or more analytical tools, understanding the assumptions made by each of them, and knowing the areas of agreement and disagreement. If multiple analytical techniques based on different assumptions and methods all point to the same locations and habitat conditions as being important factors limiting fish production, considerably more confidence can be placed in these conclusions. Application of multiple analytical procedures in the stressful context of subbasin assessment and planning may seem onerous to those required to complete these processes. Nonetheless, use of several approaches is one of the best methods of identifying the relationship between watershed condition and salmon recovery, and will help ensure that planned restoration actions address habitat conditions most likely to influence fish production. However, we emphasize that identification of useful analytical tools will require carefully designed comparisons of model outputs with actual habitat and fish population data.

Implementation Incentives

Once potential restoration locations and implementation strategies have been identified and prioritized, the question becomes how to ensure that strategies are implemented and restoration objectives are achieved. The successful implementation of tributary habitat improvements rests in large part on the incentives facing those with decision authority over tributary resources. In the case of publicly owned resources – either state or federal – mechanisms exist through various pieces of legislation, administrative rules and consultation procedures to motivate and coordinate restoration actions. Use of these should be acknowledged in the subbasin planning process.

Scientific Foundation for Habitat Recovery

Understanding large spatial and temporal-scale patterns is important in subbasin planning because restoration strategies need to identify the best sites for protection as well as the best candidates for restoration – those sites that have a likelihood of becoming highly productive at some future time. In preparing this report, the ISAB was briefed by scientists who have completed subbasin assessments that have been considered successful. Although the objectives of the assessments varied, the approaches that contributed to their success shared four common themes.

First, the assessments began with a *thorough, systematic inventory of conditions within the drainage system*. Instead of gathering as much data as possible and then attempting to make sense of it all, the investigators first identified the information needed to address their questions and then set up data collection programs to acquire it.

Second, successful assessments utilized *measures of ecosystem functionality that extended beyond the stream to encompass whole watersheds*. In each situation, rule sets were established that enabled investigators to make statements about the general ecological condition or "health" of an area based on combinations of environmental factors. This step was critical because it provided the context needed to assess the significance of habitat degradation at individual sites, as well as a means of judging the potential for effective restoration at different locations.

Third, the distribution and habitat requirements of target species over the entire freshwater life cycle were overlain on spatially referenced maps of watershed condition to allow for a *landscape-scale comparison of life-history needs and habitat status*. Superimposing fish population data on landscape condition allowed investigators to identify locations within subbasins where fish populations and ecosystem processes were robust or impaired.

Fourth, successful subbasin assessments all contained *explicit strategies for habitat recovery* (e.g., rebuilding outward from core stronghold areas; reconnecting aquatic, riparian, and floodplain systems) and provided a means of forecasting future conditions based on projected human developments.

Monitoring and Evaluation

Understanding the effect of habitat conditions on salmonid population performance requires replicated observational studies or intensive research-level experiments at large spatial and temporal scales. Few evaluations of tributary habitat in the Columbia River basin have successfully adopted either approach. Monitoring has been categorized in a hierarchical sequence (Tier 1, Tier 2, or Tier 3) in the All-H Paper and repeated in the 2000 BiOp. For consistency, we discuss monitoring and make recommendations in terms of this hierarchical sequence. The three levels are: trend monitoring (Tier 1), statistical monitoring (Tier 2), and research monitoring (Tier 3). The value of research, monitoring and evaluation is greatly enhanced if these elements are integrated. The ISAB recommends that intensive watershed monitoring at selected locations be included in overall strategies for evaluating habitat improvement projects.

Tributary subbasins of the Columbia River are a complex mosaic of different land uses and ownerships.

I. Introduction

Tributary habitat improvement has been highlighted as one of the most important cornerstones for salmon recovery in the Columbia River Basin. Tributary habitat is emphasized in salmon recovery documents, the Fish and Wildlife Program and watershed planning. Many tributary habitat restoration efforts embrace the concept of returning altered streams to natural conditions. This emphasis is in contrast to many salmon recovery projects in the mainstem Columbia River that emphasize mitigation or artificial propagation as opposed to ecological restoration. In this sense, tributary habitat restoration may lead to somewhat more "normative" conditions (ISG 2000) than those produced by recovery actions addressing the other Hs (hatcheries, harvest, and hydroelectric operations).

Whether tributary habitat improvements have achieved, or are likely to achieve, the goal of recovering conditions favoring natural production of native salmonids in the Columbia River

Basin is an open question. As recommended in the 2000 FCRPS BiOp and All-H Paper (National Oceanic and Atmospheric Administration, National Marine Fisheries Service 2000), federal and state action agencies have committed to aggressive tributary habitat restoration partly in lieu of breaching the lower Snake River dams. Tributary restoration is to be coordinated through subbasin planning (www.nwcouncil.org/fw/subbasinplanning) to ensure that habitat recovery actions are complementary to the other Hs. NOAA Fisheries has indicated it will accept subbasin planning as a process for developing recovery plans for listed Pacific salmon ESUs in the Columbia River Basin (letter from Bob Lohn to Larry Cassidy: May 24,2002).

A review of the approaches, assessment procedures, and implementation strategies for tributary habitat improvement is therefore timely. In this report we examine several topics central to the recovery of tributary habitat under subbasin planning: (1) subbasin plan objectives related to habitat recovery; (2) strategies for implementing restoration; (3) incentives for implementing restoration; (4) scientific foundation for habitat recovery; and (5) monitoring and evaluation. The overall objective of the review is to answer the question: *What concepts and strategies should be incorporated in habitat recovery actions to improve their chances for success?*

Status of Subbasin Planning

The subbasin planning process is coordinated by the Northwest Power Planning Council.

> "In 2000, the Council adopted a set of amendments to the Program to begin what will eventually be a complete revision of the Program. In the first phase of the amendment process, completed in 2000, the Council reorganized the Program around a comprehensive framework of scientific and policy principles. The fundamental elements of the Program as revised are the vision, which describes desired accomplishments regarding fish and wildlife; basinwide biological objectives, which describe physical and biological changes needed to achieve the vision, consistent with the scientific principles; implementation strategies, which will guide or describe the actions needed to achieve the desired ecological conditions; and a scientific foundation, which links these elements and explains why the Council believes certain kinds of actions should result in desired habitat conditions and why these conditions should improve fish and wildlife populations in the desired way.
>
> The Program amendments in 2000 set the stage for subsequent phases of the Program revision process, in which the Council will adopt more specific objectives and measures for the tributary subbasins, consistent with the framework elements already adopted. The Council intends to incorporate these specific objectives and measures into the program in locally developed subbasin plans for the 62 subbasins of the Columbia River (along with a coordinated plan for the mainstem Columbia and Snake rivers). The subbasin plans will become the source of specific actions and projects recommended by the Council for Bonneville funding and implementation, and will provide the context for the review of proposals for funding by the Council and the Independent Scientific Review Panel."

The basic elements of a subbasin plan are described in a recent posting on the Council's website: (www.nwcouncil.org/fw/subbasinplanning/admin/recommendations.htm)

> "Any subbasin plan adopted into the Program must consist of three general components:
>
> - A subbasin assessment providing a description of historical and existing conditions;
> - A clear and comprehensive inventory of existing projects and past accomplishments;
> - A 10-15 year management plan with a vision, biological objectives and strategies for the subbasin."
>
> "…all recommendations for subbasin plans [are to] be submitted to the Council on or before Friday, May 28, 2004. This deadline seeks to provide the maximum amount of time available for developing recommendations while allowing for a Council amendment proceeding to adopt plans by the end of 2004."

Before subbasin plans are formally submitted to the Council as recommendations, the Council has offered scientific review of final drafts subbasin plans. This will allow subbasin planners to receive input from the science review and make any changes that they decide are appropriate before submitting a final formal subbasin plan recommendation to the Council. The Independent Scientific Review Panel (ISRP) will conduct the reviews, during which they will answer the following questions:

> "Scientific review will evaluate proposed subbasin plans for their consistency with the Scientific Foundation adopted as part of the Program and with the requirements for "biological objectives" as described in the program. Scientific review will evaluate whether proposed plans are 1) internally consistent and 2) scientifically sound. Internal consistency means there is scientific support for the conclusion that the strategies proposed for a subbasin plan will in fact address the problems identified by the subbasin assessment. In evaluating whether subbasin plans are scientifically sound, the scientific review will be guided by the following considerations:
>
> - Do the assessments appear to be thorough and substantially complete?
> - Are the subbasin goals, objectives, and strategies scientifically appropriate in light of the assessment and inventory of existing activities?
> - Does the plan demonstrate a linkage between the strategies, the biological objectives, the subbasin vision and the assessment?
> - Are the goals, objectives, and strategies consistent with those adopted in the program for the province and/or basin levels?
> - Do the plans demonstrate that alternate management responses have been adequately considered?
> - Does the proposed subbasin plan include a procedure for assessing how well subbasin objectives are being met over time?

- Does the plan provide a scientifically supportable procedure for refining the biological objectives as new information becomes available about how fish, wildlife and the environment interact, and in relationship to how the plans are implemented over time?

Once the subbasin plans are formally submitted, the Council will conduct its own review, which will include Council staff and public review. The Council staff will also facilitate analysis of the subbasin plans using the Ecosystem Diagnosis and Treatment tool, allowing the Council to evaluate and understand in a consistent way how different plans have incorporated assumptions about how fish and wildlife are affected by their environments.

The Columbia Basin Fish and Wildlife Authority (CBFWA) has completed reviews of each of the Columbia River Basin's 11 geographic provinces, some of which align, more or less, with ESU boundaries, as part of the rolling review process for Fish & Wildlife Program Funding (www.cbfwa.org/province.htm). Included in the provincial reviews are assessments of the habitat conditions and fish populations in major tributary systems. This information will be incorporated into subbasin assessments and the inventory of existing and past projects. The NPPC has applied EDT to many of the tributaries in the Columbia Basin (www.edthome.org/Default.htm) for the purpose of assessing current habitat conditions at the sub-watershed (HUC-6) level, and for the purpose of comparing the model's estimate of current productivity to what the sub-watershed could support under "best case" conditions. For an example of EDT output, see the model's results for chinook salmon in the Chiwawa River, a tributary of the Wenatchee River subbasin at www.edthome.org/level2/viewer.asp?from=subbasinplanning&fldWatershed=Wenatchee.

To date, "subbasin plan workplans" have been approved for a majority of the Columbia River Basin subbasins, e.g., the Flathead River subbasin workplan (www.nwcouncil.org/fw/subbasinplanning/flathead/workplan.pdf). This workplan provides a roadmap for developing the full subbasin plan for the Flathead River system over a 14-month period, at a cost of approximately $150K. The Flathead workplan is to serve as the model for other subbasin plan workplans in the Columbia River Basin. Based on the Council's schedule, all of the subbasin plans will be completed by 2005, in time for the 5-year interim review prescribed in the BiOp.

The first completed subbasin plan, for Idaho's Clearwater River, was submitted to the Council on November 13, 2002 (www.nwcouncil.org/fw/subbasinplanning/clearwater/default.asp). The Clearwater plan was developed by the Clearwater Policy Advisory Committee, which included representatives of Idaho County, Potlatch Corporation, Nez Perce National Forest, the Nez Perce Tribe, U.S. Fish and Wildlife Service, Idaho Department of Environmental Quality, Idaho Department of Fish and Game, Idaho Department of Lands, Idaho Association of Conservation Districts, Clearwater National Forest, and NOAA Fisheries. The Independent Scientific Review Panel completed its review of the Clearwater subbasin plan on February 18, 2003 (www.nwcouncil.org/library/isrp/isrp2003-3.htm). The Clearwater planners are now reviewing the ISRP report and will make any changes that they decide are appropriate before submitting a final formal subbasin plan recommendation to the Council. The Council then will conduct its own review and undertake a formal rulemaking process, including a public comment period, prior to adopting the plan.

Tributary Habitat Restoration Framework

An overview of the procedure for developing subbasin plans for tributary habitat recovery is given at www.nwcouncil.org/fw/subbasinplanning/admin/guides/overview.htm. There are six steps in the process:

1. Assessment

An assessment forms the foundation for developing the subbasin vision, biological objectives and strategies. The initial assessment is based on existing information about the environmental conditions and fish and wildlife populations in the subbasin. A key element of the assessment will be information on the current and potential conditions in each subbasin. From this assessment, the subbasin plan will identify limiting factors and factors for decline for key fish and wildlife populations in the subbasin, including ESA-listed populations. Where the assessment identifies significant data gaps, the subbasin plan should identify the data need and measures necessary to meet those needs. The assessment should address the question, "What are the problems that keep fish and wildlife populations within the subbasin from reaching full potential?"

Examples of limiting factors and factors for decline

- Water quality problems in the lower river (temperature and sedimentation)
- Passage barriers at culverts and falls (late summer)
- Lack of adequate screening
- Overwinter habitat is insufficient
- Lack of juvenile rearing habitat
- Low fish or wildlife abundance
- Reduced biological function of habitat above blockages

2. Vision

The intention of the Council's subbasin planning effort is to define the environmental and biological goals specific to fish and wildlife within the Columbia River Basin. The Council anticipates a 10-15 year timeframe as the planning window. A vision statement is qualitative, and should reflect the policies, legal requirements and local needs, given the ecological realities within a subbasin. The vision will provide the guidance and priority for implementing actions in the future. The vision for the subbasin should address the question, "What are you trying to achieve overall?" -- a collective desire to accomplish certain things.

Examples of collective goals forming the vision

- Restore fish runs
- Maintain genetic integrity
- Protect and restore wildlife habitat
- Increase harvestable populations of fish
- Increase escapement to the spawning grounds
- Rebuild fish runs to achieve ESA delisting

3. Biological Objectives

Biological objectives have two components: (1) biological performance, describing responses of populations to habitat conditions, described in terms of capacity, abundance, productivity and life history diversity, and (2) environmental characteristics, which describe the environmental conditions or changes sought to achieve the desired population characteristics. Objectives should be specific, measurable, and quantifiable. The initial assessments along with the vision will guide the focus of the biological objectives. For each major limiting factor, there should be a biological objective that describes the extent of improvement that the plan will call for. In addition, for each key population, specific biological objectives should describe the improvements planned for that population. These objectives will serve as a benchmark to evaluate progress toward the subbasin vision, and should have measurable outcomes. The questions that should be addressed through the biological objectives are "What target species need to be addressed?" "What number is achievable, and in what time frame?" Immediate, interim, and long-term biological objectives should be considered.

Examples of biological objectives

- 2,700 summer steelhead return to spawn by 2006;
- 5,000 spring Chinook return to harvestable levels by 2008.
- Increase winter rearing habitat by 10%.

4. Strategies

Strategies describe the actions needed to address the limiting factors and therefore achieve the biological objectives. The strategies identified in the subbasin plans form the basis for Council funding recommendations to the Bonneville Power Administration. Implementation strategies will vary depending on the current condition of the populations and habitat, and the biological objectives identified for the species and life stages of interest. Strategies should be formulated to address the question, "What are the generic or overarching actions needed to address the limiting factors?"

Examples of strategies

- Improve water quality in the lower river
- Restore passage through a particular barrier
- Restore riparian habitat in a particular stream reach

Strategies will be implemented through specific projects and/or actions. Projects proposed for funding will not be identified within the subbasin plan. When a plan is approved, it will form the basis for project selection within the subbasin. Projects will be developed through the regional project funding process. Projects proposed for funding will undergo independent scientific review as to how they fulfill the strategies and biological objectives in the subbasin plan.

An example of a strategy with related projects

Strategy
Projects (submitted through province review)

Restore fish
Build a fishway at Sunny Creek

Passage
Increase instream flows - upgrade Sunny Farm diversion

5. Research, Monitoring, and Evaluation

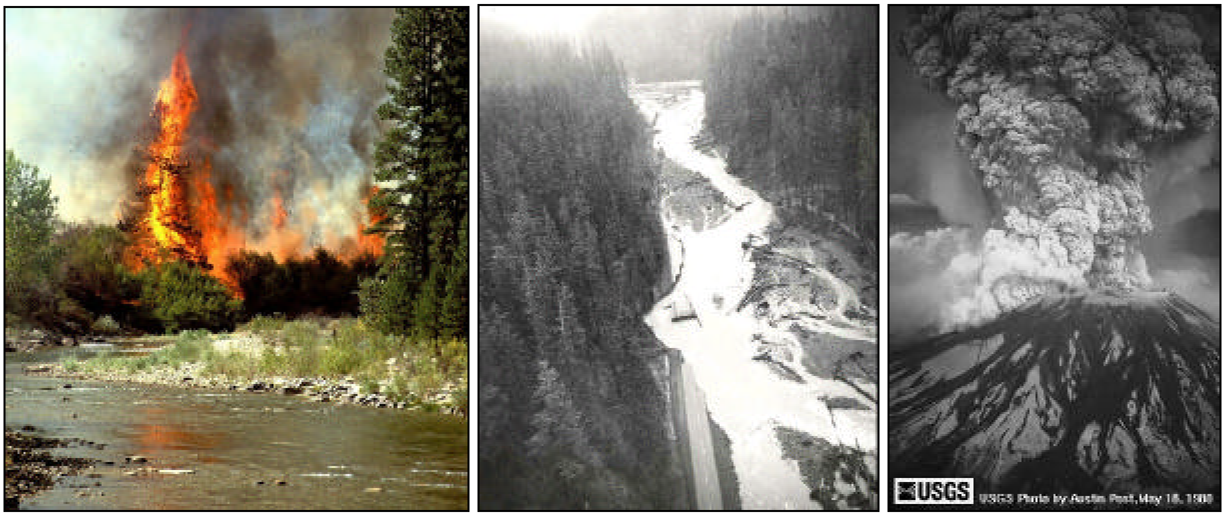
Each subbasin plan will contain a monitoring and evaluation plan that will show whether the actions taken to implement the subbasin plan are achieving their objectives. Each monitoring and evaluation plan should answer the questions "How will we evaluate progress toward the biological objectives?" "How will it be measured?" "Who will conduct the monitoring and evaluation work?" and “What is the timeframe for such work?" The information gained through monitoring and evaluation allows for the examination of the effectiveness of actions taken so that actions may be refined over time.

In addition, each subbasin plan will contain a set of research questions (agenda) that will address critical uncertainties related to stated goals, biological objectives, and strategies that will become part of a larger research plan for the basin. The research agenda recognizes conditions and situations identified within a subbasin that will require specific research in order to help resolve specific management uncertainties.

6. Appendices

The background information and supporting documentation used in subbasin plan development can be included as technical appendices to the plan. Components of the technical appendices should include:

- Assessment and limiting factors data and information;
- Project listings and summaries -- inventory of existing projects, program and past accomplishments;
- Subbasin summaries developed for the Council;
- Maps, excerpts, and other relevant documents.



Wildfires, floods, and even volcanic eruptions are examples of natural disturbances that often cause short-term damage, but are needed to sustain complex aquatic habitats and long-term productivity. Accommodating natural disturbances often conflicts with the notion of fixed habitat standards.

II. Establishing Biological Objectives

The vision for each subbasin will be articulated by local, state, tribal, and federal policy-makers, addressing each of the Hs, within the context of the overarching mandates for fish and wildlife recovery in the Columbia River Basin. Subbasin plans are expected to contain clear statements of the biological objectives for each major tributary system, including population recovery targets for key species and expected improvements in habitat. The biological objectives developed in each subbasin plan are important because they provide measurable targets for habitat recovery. Biological objectives for habitat will emerge from four major considerations:

- Tributary habitat performance standards
- Limiting factor determination
- Definition of the "habitat template" (in EDT, for example, the habitat template is defined as the "hypothetical potential state where conditions are as good as they can be within the watershed")
- Association of fish populations with habitat conditions

These four issues are closely related. Tributary performance standards are referenced in the All-H Report (Vol. 2, pp. 13-14) and are taken to mean the specific habitat states that would achieve biological recovery in an area of interest. These standards become the *de facto* biological objectives for tributary habitat. Development of realistic and achievable tributary habitat performance standards is crucial; without them, prioritization of restoration efforts is unclear and prediction of population response to habitat restoration will be highly uncertain.

a. Tributary habitat performance standards

What are the merits of fixed habitat targets (e.g., TMDLs, minimum dissolved oxygen concentrations, maximum stream temperature and sediment levels), as opposed to managing for the "distribution of natural conditions"?

Habitat standards have been with us for decades, especially since enactment of the Clean Water Act in 1972. They usually appear in regulations as, for example, maximum allowable water temperature or minimum levels of dissolved oxygen. Environmental targets, e.g., the number of pieces of large wood per unit length of stream or minimum instream flows, are used as a basis for habitat restoration decisions and to justify land and water use restrictions. In many management schemes, these standards serve as indicators of fish habitat condition and as barometers of management performance (USDA Forest Service/USDI Bureau of Land Management 1995).

In many cases the application of environmental standards and performance thresholds will divert attention from the real issue – managing watersheds in such a way that ecological processes supporting aquatic productivity and diversity are restored and conserved. Habitat standards have often failed to protect salmon because they are taken as fixed and do not focus on dynamic processes that create and maintain ecologically complex and resilient watersheds (Reeves et al. 1995; Bisson et al. 1997). Fixed habitat standards do not easily account for environmental variations, differences in habitat requirements among species, or temporal changes in habitat requirements over a fishes' life cycle. Additionally, managers often treat standards as targets, tolerating habitat degradation until a danger threshold is exceeded and only then taking some corrective action. This approach is inappropriate because the general trend is to homogenize habitat rather than maintain the complexity of conditions that support biological diversity at multiple scales. In addition, applying generic assumptions about environmental thresholds, often obtained from laboratory studies, can lead to unrealistic habitat standards (Shirvell 1989).

Variation in environmental conditions has been identified as an important component of their habitat requirements for some species of wildlife. Erickson et al. (2003) studied habitat selection by alder flycatchers (*Empidonax alnorum*) on the Innoko National Wildlife Refuge in west central Alaska. Locations of breeding alder flycatchers were gathered by walking systematically-located transects, orientated perpendicular to the river channel, within the flood plain of the Innoko River during the 2000 and 2001 breeding seasons. Data from these surveys were used to construct a habitat selection function to predict the likelihood of the presence of alder flycatchers from habitat condition (Manly et al. 2002). Candidate predictor variables included both physical site descriptors such as slope, aspect, elevation, and distance to the river, and vegetation characteristics inferred from LandSat imagery. The variables with the highest importance in predicting the presence of the birds were elevation and two LandSat-derived attributes: the band 4 value and standard deviation of the band 3 value. In this example it appeared that the birds were responding to both the average condition of the site and the degree of variation in habitat conditions.

Productive habitat for fish and wildlife exhibits complex structural diversity in space and time. Populations persist under these variable conditions because they have a complex structure of sub-populations, some strong and some vulnerable, distributed across a wide array of habitats.

Extirpation in one area can be compensated, in time, by emigration from an adjacent sub-population. Similarly, low production in one area may be compensated by above average production in adjacent areas. Scientists can only make educated guesses regarding the optimal population structure and habitat patterns for a successful population. Projects to improve watershed processes that produce productive natural habitat for fish and wildlife probably are beneficial in most situations, but by themselves, are likely to make only minor contributions to restoration of the structure needed by a successful population of wildlife or fish. We recommend that attention be focused on identifying, as soon as possible, the overall spatial array of watersheds and habitat units needed to protect important populations.

Habitat standards have been used in modeling systems. However, major sources of error can arise from modeling fish distribution and abundance from a limited number of physical habitat variables without considering the full array of ecological interactions that are potentially controlling factors. For example, the Instream Flow Incremental Methodology (IFIM) has been widely employed to estimate usable rearing habitat at different discharge levels (Bovee and Cochnauer 1977; Milhouse et al. 1981). IFIM modeling is often used to establish minimum instream flows in connection with the management of regulated rivers, despite objections to its accuracy if the model is not adequately calibrated with local biophysical data (Annear and Conder 1984; Mathur et al. 1985; Kondolf et al. 2000). Habitat-based population models can be useful, but may not yield accurate predictions about population responses to ecosystem change unless parameterized with data from populations in question (Fausch et al. 1988).

Proponents of environmental standards often regard spatial and temporal variation as a sampling problem and ignore variation in habitat requirements within or among species. An alternative view is that variation is an important part of the spatial and temporal landscape that deserves a place in environmental planning. Natural disturbances create spatial and temporal variability essential for maintaining the distribution of habitat conditions necessary to support high levels of aquatic productivity and diversity (Reeves et al. 2002). Large natural disturbances such as wildfires and floods often result in short-term habitat loss and population stresses (e.g., through inputs of fine sediment or spawning gravel scouring) but over longer cycles of disturbance and recovery are important contributors of wood, coarse sediment, and other structural roughness elements to streams. As streams recover from a disturbance, habitat conditions change and biological communities utilizing the stream also change. Disturbance itself may be directly important as part of the environmental template for evolution. The maintenance of genetic and phenotypic variation and plasticity in life history characteristics that allow populations to persist in the face of natural and anthropogenic disruption of habitats may depend on habitats naturally varying in space and time (Poff and Ward 1990; Reice et al. 1990). Large natural disturbances create complex mosaics of aquatic and floodplain habitats necessary for different fish life history stages. They also create a full range of environmental conditions needed to maintain biodiversity at spatial scales ranging from the stream reach to the province.

The short-term impact of wildfire often includes increased fine sediment and elevated stream temperature, but ecological recovery can be rapid and the long-term benefits of fire include recruitment of large wood and coarse sediment for habitat formation.

Suppose a habitat performance standard was determined to be the percent of old growth coniferous forest. Old growth forests are known to be important for fish and wildlife, and the percent of old growth remaining in the Columbia River Basin is now far below historic average levels, so this might be a reasonable choice. What were historical levels and how could we set old growth landscape targets? Some of the best work has been done in the Pacific coastal ecosystem in the range of the northern spotted owl, an old growth-associated species. Vegetation specialists, climatologists, and geographers have attempted to reconstruct the percentage of old growth forests in the Oregon Coast Range using disturbance simulation models (Wimberly et al. 2000). Their findings are relevant to the question of applying standards at different geographic scales (Figure 1).

In Figure 1, the upper graph illustrates long-term variation in the percentage of old growth forest at the province scale beginning at present and extending back 3,000 years. The average percentage of old growth at 500-yr intervals ranges from about 40% to 55%. However, the middle and lower graphs reveal an increase in temporal variability with successively smaller landscape units. Most of this simulated variability is due to large wildfires, forest disease outbreaks, and climate change. Wimberly et al. (2000) conclude "Until we can estimate ranges of historical landscape variability more accurately, it will be difficult to substantiate an argument for their use as precise forest management goals".

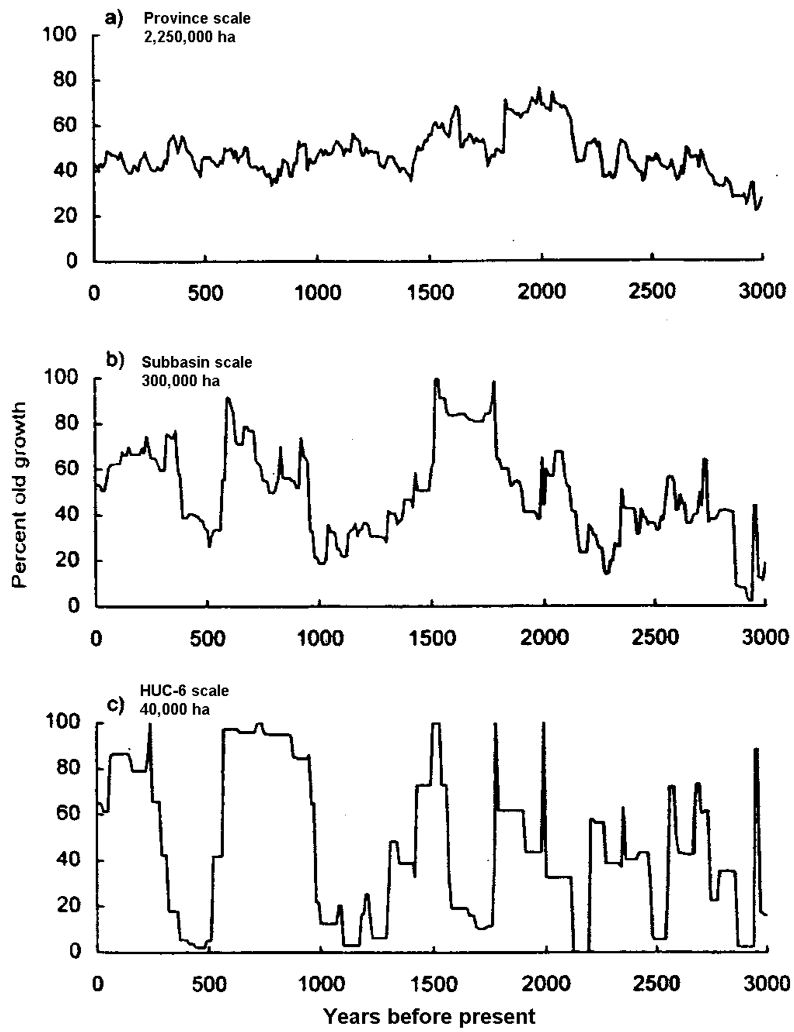


Figure 1. Percent of old growth coniferous forest over the past 3,000 years in the Oregon Coast Range based on simulation modeling. From Wimberly et al. (2000) reprinted with permission from the Society for Conservation Biology.

Although this analysis is a computer simulation of the Oregon Coast Range, it shows how much landscape conditions can vary under natural disturbance regimes. There is no reason to believe many areas of the Columbia River Basin have been any less variable. Attempting to define an old growth habitat performance standard, especially at the subbasin and smaller scale, becomes problematic because natural variation is so extreme over periods of decades to centuries as to make average conditions essentially meaningless. Similar patterns of variability are likely among many other environmental attributes.

Graph (a) in Figure 1 suggests some habitat parameters could vary within somewhat narrow limits at very large scales (province) and it is at this scale that applying standards makes the most sense. But even at this large scale, setting a standard based on an average condition over time does not acknowledge the inherent natural variation. As a result, this type of standard only reflects "natural conditions" that occur for a very limited period of time. Using such a standard could be counterproductive to achieving biological objectives if temporal variation is important to biological productivity and diversity. In addition, targets generalized over very large landscapes and time scales may not be very helpful at the site level where habitat management decisions are most often made. How do we know if departure from desired conditions at the scale of a watershed, subwatershed, or stream reach merits restoration? Can generalized landscape target conditions be applied at the level of a subwatershed or stream reach?

Some environmental attributes are simply too variable at small spatial scales to be used as habitat standards or biological objectives. Two of the best examples are stream temperature and fine sediment concentration. Temperature varies along stream gradients from headwaters to mouth, due primarily to altitudinal changes in air temperature and increases in direct solar radiation as streams widen and both riparian and topographic shading decrease. Diel temperature change can be considerable where long-wave radiation loss cools stream water at night (Beschta et al. 1987). Inputs of ground water or hyporheic flows create localized cool water areas in what might otherwise be warm streams in summer (Bilby 1984). At any point in the drainage network or at any time of day, temperature can be highly variable, an exception being when flow is largely controlled by reservoir releases. Forward Looking Infrared (FLIR) surveys in the John Day River Basin has revealed numerous cool water zones in the mainstem associated with small tributaries and groundwater seeps (Bruce McIntosh, Oregon State University, personal communication). Salmonids have been shown to utilize pockets of cool water in Columbia Basin tributaries, enabling them to maintain a body temperature as much as 10^{o}C lower than the average water temperature (Berman and Quinn 1991). Given the spatial and temporal variation in stream temperature and the possibility of behavioral thermoregulation by salmon, the biological meaning of a single-value temperature standard for protecting salmon habitat is questionable.

Likewise, fine sediment deposits in stream channels are highly variable. There tends to be a gradual increase in deposited fine sediment in a downstream direction due to lessening gradients, but actual fine sediment levels at a particular site are dictated by local soils and geologic rock type, the abundance of large roughness elements such as large woody debris that retain sediment, geomorphic irregularities in the channel, inputs from natural and anthropogenic disturbances, and stream power (Everest et al. 1987; Megahan et al. 1992). At any given location there is often a wide range in fine sediment concentration within the stream channel that makes characterization of average conditions very difficult and not biologically meaningful.

The spatial patchiness and temporal variability of both temperature and fine sediment requires that measurement of either of these parameters be undertaken with careful consideration of environmental heterogeneity. At present we are aware of no standardized sampling protocols for stream temperature that capture this heterogeneity in a way that can be interpreted ecologically. Even a thorough review of fine sediment in streams (Chapman 1988) concluded that relating fine sediment concentration to salmonid spawning success requires intimate knowledge of spawning locations so samples can be obtained from egg pockets, and further, that interpretation of resulting data be related to sediment tolerances of different species. Without this information, estimation of the effects of fine sediment on egg and alevin survival will be speculative.

In the 15 years since Chapman's review little has been done to improve sediment sample methods or to refine our understanding of sediment effects on salmon and trout. Some promising new technologies have been applied to temperature, including Forward Looking Infrared surveys. We conclude that sampling methods for temperature and sediment are currently too poorly developed to adequately characterize the variability in these parameters. Therefore, it is unlikely that standards at the scale of watersheds, subwatersheds, or stream reaches would be useful and it is highly doubtful they would be of much use at the province level.

This same conclusion may well apply to other physical parameters associated with stream habitat (Poole et al. 1997). Standards are not available for many ecological attributes, and even in cases where standards have been established, their biological relevance may be questionable. Rigid performance standards for physical system attributes fail to account for the complex array of habitats required to support biological diversity at multiple scales of space and time; simple standards cannot encompass the dynamic processes that create and maintain ecologically complex and resilient watersheds (Bisson et al. 1997). Using fixed habitat standards and thresholds to assess current watershed condition and evaluate progress of restoration strategies potentially diverts attention from the primary objective of most restoration efforts – re-establishment of processes that support aquatic productivity and diversity.

Habitat standards based on the distribution of conditions observed in unmanaged landscapes are more ecologically relevant if they are expressed at appropriately large scales of space and time. Large spatial and temporal scales are required to account for the variety of conditions generated by natural processes of disturbance and recovery in unmanaged landscapes (Reeves et al. 1995; Bisson et al. 1997). In many cases, decades or centuries might be required for habitat attributes within a degraded watershed to achieve a range of conditions comparable to those exhibited by an unmanaged system. Therefore, performance standards may be more usefully articulated by coupling the potential range of parameter conditions (i.e., median, range, and variance) with a predicted rate of change from the current to the desired state. Standards expressed in this manner have the advantage of providing meaningful feedback on the efficacy of management actions much faster than simple comparison to a fixed set of habitat standards.

"Distribution of natural conditions" standards require an understanding of conditions in watersheds with little human impact or watersheds with high-levels of natural production for the species of interest. Because unimpacted watersheds no longer exist in many areas, determining a natural range of conditions for many ecological attributes is often problematic. Two approaches to address this deficiency have recently been applied:

- Modeling tools that predict distributions of conditions for some watershed attributes, such as forest stand ages or large wood in stream channels (Benda et al. 1998; Wimberley et al. 2000). These tools may prove useful for establishing desired outcomes for landscapes where no adequate reference information is available.
- Historical information from a variety of sources to understand the range of variability in certain watershed conditions (Collins and Montgomery 2001). These historical reconstructions have been developed for a number of Puget Sound watersheds (Collins et al. 2003).

In most cases, establishing desired distributions of conditions may be accomplished by linking empirical information from unmanaged watersheds to historical data using models that enable projections of the range of previous watershed conditions. An understanding of the range of conditions of a watershed through time provides a basis for establishing desired trends in monitored parameters in watersheds where habitat degradation has occurred. For example, maximum summer water temperatures recorded at multiple locations in an unimpacted or highly productive watershed might exhibit a frequency distribution as shown in Figure 2. If the distribution of water temperatures in a nearby degraded or low productivity watershed deviates significantly from the distribution of maximum temperatures in the unimpacted watershed, the desired outcome of a restoration program to address water temperature would be a gradual shift in this frequency distribution toward that observed in watersheds with the desired biological attributes. This approach acknowledges that a proportion of streams in a productive landscape will exhibit water temperatures different than those considered desirable for certain species. Natural temperature deviations are usually caused by disturbances that remove streamside vegetation, such as wildfires or floods. These periodic disturbances play a key role in maintaining long-term productivity of aquatic systems and are a vital process for maintaining watershed health (Bisson et al. 1997). Thus, warm water at some sites is not ecologically undesirable, and in fact may be needed for native species with higher temperature preferences than salmonids. Restoring an appropriate distribution of water temperatures in a developed watershed would represent a management goal more realistic and biologically meaningful than a single, fixed water temperature value.

Stream channels are highly variable. Understanding the distribution of natural conditions in relatively unaltered watersheds is important for establishing realistic habitat goals and trends.

An understanding of the distribution of natural conditions of a watershed through time provides a basis for establishing desired trends in monitored parameters in watersheds where habitat degradation has occurred. For example, maximum summer water temperatures recorded at multiple locations in an unimpacted or highly productive watershed might exhibit a frequency distribution as shown in Figure 2. If the distribution of water temperatures in a nearby degraded or low productivity watershed deviates significantly from the distribution of maximum temperatures in the unimpacted watershed, the desired outcome of a restoration program to address water temperature would be a gradual shift in this frequency distribution toward that observed in watersheds with the desired biological attributes. This approach acknowledges that a proportion of streams in a productive landscape will exhibit water temperatures different than those considered desirable for certain species. Desired distributions for various habitat parameters can be articulated in this manner. Wood abundance, pool frequencies, and other habitat attributes all exhibit spatial and temporal variation. This natural variation is caused by local differences in underlying physical features of the landscape, disturbance history, and dynamics of recovery processes. Attempts to restore most of these attributes will require long time periods to achieve the desired conditions. Evaluation of the effectiveness of a restoration effort may be more rapidly judged by tracking the direction and rate of movement of the distribution of conditions in a watershed towards the desired median, range, and variance.

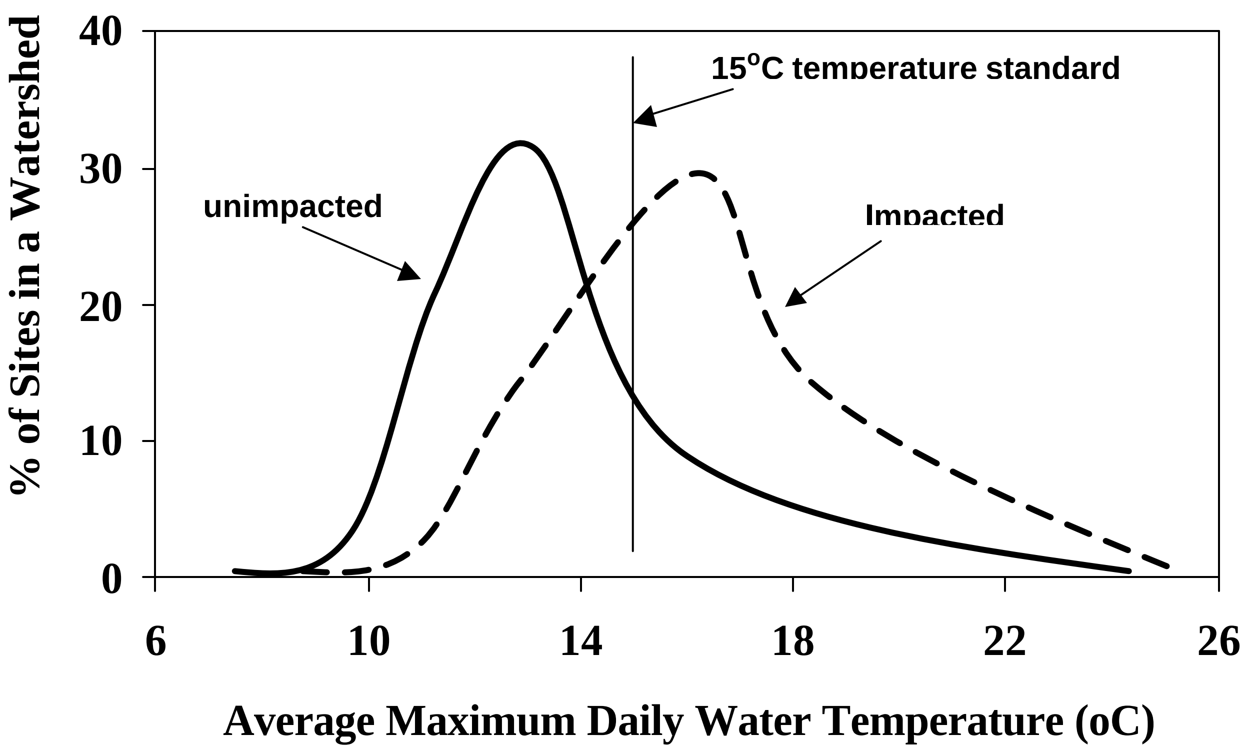


Figure 2. Theoretical frequency distribution of average maximum daily water temperature in an impacted and an unimpacted watershed.

An illustration of this approach is provided by the following example. The data to estimate the natural distribution of large wood in forested streams were collected by Fox (2001). Wood in streams selected at random from areas of unlogged forest ranged widely in abundance (Figure 3). Even after accounting for the effects of channel size and forest type, natural wood loading varied by about a factor of 10. By measuring wood abundance (or a surrogate of wood abundance such as riparian forest age) at multiple sites in a managed landscape over time, the rate at which wood abundance in a restored watershed approaches the distribution of wood in a comparable reference watershed could be estimated. The evaluation could be further refined by developing an expected rate of change in riparian conditions or wood abundance using one of several predictive models for instream habitat that have recently been developed (Beechie et al. 2001; Welty et al. 2002).

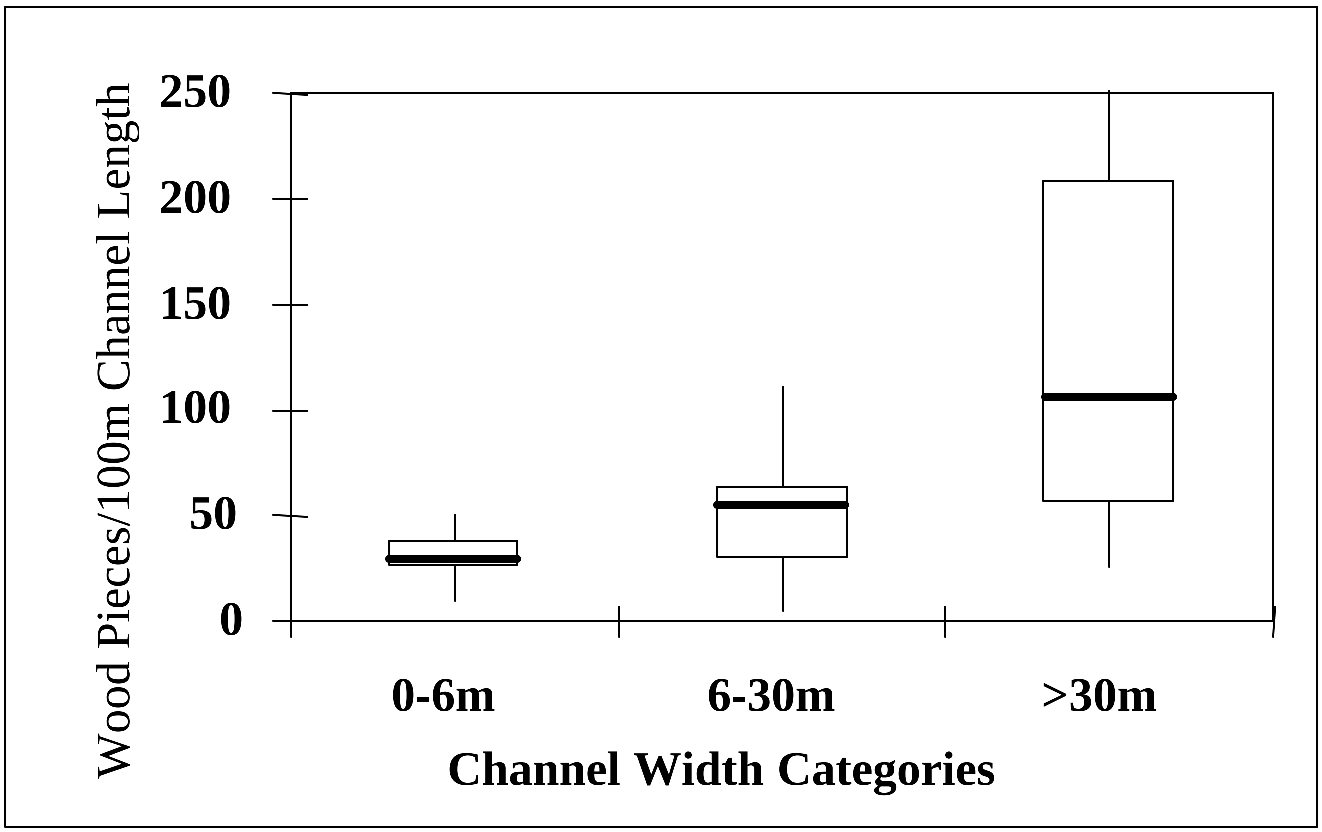


Figure 3. Distribution of wood abundance in streams flowing through unmanaged forests in the western Washington Cascade Mountains of Washington State. Values are the median abundance (dark line) the 25th and 75th percentiles of the distribution (box) and the 5th and 95th percentile of the distribution (whiskers). Data from Fox (2001).

Can habitat targets be related to the abundance or diversity of fishes?

The high mobility of fishes, especially anadromous fishes, and the great degree of variation in population size they display over space and time complicates the process of evaluating their response to changes in aquatic habitat. Some of this variation is not related directly to freshwater habitat quality but to the variable effects of weather and flow conditions on survival and growth, or to factors impacting fish in the marine environment that affect the number of adults returning to spawn. Because this variability is not a direct product of the condition of freshwater habitat, it is difficult to account for when attempting to relate fish abundance to habitat condition.

The relationship between freshwater habitat and productivity of fish populations has traditionally been examined at very fine spatial scales (individual habitat units or short stream reaches) over short periods of time (one to five years). The goal of much of this research has been to establish an association between an environmental factor and a life-stage specific response, such as the effect of fine sediment on the survival of incubating eggs (Everest et al. 1987) or the relationship between the amount of pool habitat and densities of juvenile salmonids during summer or winter rearing periods (Reeves et al. 1989). Although these studies are important for clarifying the mechanisms by which various factors affect salmon populations, they cannot provide meaningful information about the broader biological consequences of a restoration action unless they are part of a large scale, integrated evaluation effort.

Quantification of biological responses to restoration actions that affect the quality, quantity, or distribution of aquatic habitats is complicated by the diversity of habitat types required by salmon and trout to complete freshwater rearing (Table 1). The relative availability and distribution of the numerous habitat types required over the entire freshwater life cycle plays a key role in determining the survival rate and productivity of a fish population. Habitat-population relationships are further complicated as the relative importance of each habitat type can change from year-to-year due to variations in weather, abundance of fish spawning within the watershed and other factors. For example, smolt production can be dictated by spawning habitat availability and quality during years when scouring flood flows occur during egg incubation (Hartman and Scrivener 1990). However, when flows are moderate during egg incubation, population performance may be more influenced by the availability of food during spring and summer or adequate winter habitat.

Table 1. Changes in the habitat requirements of coho salmon during freshwater rearing. The changing requirements of the fish stress the need to develop monitoring designs that evaluate responses at a spatial scale large enough to encompass the full range of habitat types required by the fish to complete freshwater rearing.

Life History Stage	Habitat
Spawning and egg incubation	Gravel bedded riffles and pool tail outs in proximity of cover suitable for adult spawners (e.g., deep pools, undercut banks, debris jams)
Early fry rearing	Low velocity with cover in close proximity to food source typically associated with shallow, channel margin habitat with cover from wood and overhanging vegetation
Summer rearing	Pool habitat with cover in close proximity to food source typically associated with low gradient channels, pool/riffle morphology, streams in flood plain valley type
Winter rearing	Low velocity refuge with cover typically associated with off-channel habitat on floodplains including low gradient tributaries, secondary channels and ponds

The scale at which biological evaluations of restoration projects are conducted is dependent on the nature of the restoration effort and the biological responses being evaluated. Some biological responses can be evaluated at the reach level and related to application of a single restoration action. For example, the biological effects of an action designed to reduce the input of sediment at a road crossing can be evaluated by comparing sediment levels in spawning gravel and egg survival upstream and downstream of the road before and after implementation of the corrective action.

The ultimate effectiveness of a restoration program is reflected by the performance of the fish through their entire period of freshwater residency. Therefore, performance must be evaluated at a scale sufficiently large to enable that species to complete freshwater rearing. There are numerous measures that can indicate biological response at this scale. For example, survival can be estimated by comparing the number of eggs deposited in a watershed to the number of smolts from that cohort that ultimately leave the watershed. These Tier 2 watershed-level assessments (2000 BiOp) ideally are coupled with evaluations of effects of Tier 1 site-level restoration projects on specific habitat types and life stages. Information collected at nested spatial scales provides both an indication of the cumulative success of all projects within a restoration program and relative efficacy of individual projects.

What is the risk that setting habitat targets favorable for one species might result in unfavorable conditions for other species?

In most cases, restoring natural features of streams and adjacent riparian areas will be sufficient to provide the physical template that supports productive aquatic ecosystems. However, in some situations habitat targets may be set to specifically enhance one or two species of interest. Such targets typically include percentages of pool habitat, levels of fine sediment, pieces of large wood, or the amount of overhead cover, and are based on laboratory or field studies showing that the target conditions were optimal for the species in question. But habitat requirements differ by species and life stage, and what may be optimal for one species at one time may be distinctly sub-optimal for others. What is the risk that setting habitat restoration goals based on a few key species will lead to a reduction of other, less commercially, recreationally, or culturally important organisms?

We are aware of no studies of habitat recovery in western North America in which significant losses of biodiversity occurred after restoration, and in some case studies, restoration of ecological function has led to development of more diverse aquatic communities (Frissell and Ralph 1998). However, there are numerous warnings in the ecological literature about the pitfalls of attempting to optimize stream habitat for certain species where such attempts run counter to natural stream conditions and the maintenance of natural species assemblages (Sedell and Beschta 1991). The following quotation from the National Research Council (1992) expresses a scientific consensus that attempting to optimize streams for target species often leads to unanticipated and often undesirable outcomes:

> "Some fisheries biologists believe that 'water and space are going to waste' if they are not used by trout and that 'even the best streams could be made better' by producing more trout in them. To the ecologist interested in stream or river restoration, maximizing the ecosystem for trout, or any single species, is not the same as restoring the biotic structure and function of the stream [p. 229]"

There is a danger, of course, that attempting to restore stream conditions favorable for salmonid production will produce harmful effects on non-salmonid species. For the most part, this has not been conclusively demonstrated in the Columbia River Basin, in part because non-salmonid species are rarely monitored. Although the ISAB doubts that this will be an issue in most situations, we do acknowledge that the habitat requirements of some native species are quite different from the habitat needs of salmon and trout. Lamprey (*Lampetra* spp.), for example, spend the early years of their life cycles burrowing through fine grained substrates in search of fine particulate organic matter upon which they feed. These sediment and organic matter-rich stream substrates are needed by lamprey ammocoetes, but sand- and silt-dominated substrates are considered undesirable for salmonid production. Imposing habitat targets that would lead to significant reductions in fines in stream gravels, while undoubtedly beneficial for salmonids, would probably reduce habitat suitable for juvenile lampreys. Likewise, setting habitat targets that call for reducing stream temperatures may benefit juvenile salmon at the expense of cyprinid and catostomid fishes that prefer slightly warmer water (Reeves et al. 1998). Such targets may be viewed as desirable by managers, but if a native non-salmonid were to be listed under the Endangered Species Act, habitat conflicts could arise.


Installing large wood in streams, excluding livestock, and planting trees in riparian zones (protected by plastic mesh tubes in the right photograph) are common techniques of active restoration. Active restoration projects are usually designed to address factors that are believed to limit natural productivity.

b. Limiting factor determination

Limiting factors are usually taken to mean those factors that constrain the production of a species of interest. Typically cited examples of limiting factors for stream-dwelling salmonids include high temperature, excessive levels of fine sediment in spawning gravels, and lack of suitable winter habitat. By implication, “fixing” these environmental factors will “lift the lid” on production and allow target populations to attain higher sustainable growth and abundance.

What is the basis for believing that habitat factors limit salmonid populations in Columbia River Basin streams (i.e., lab studies, field experiments, research with closely-related species)?

The usual procedure for identifying limiting factors is to compare current conditions with published data on environmental tolerance. For example, if eggs of a particular species are known to undergo significant mortality when exposed to dissolved oxygen concentrations of less than 5.0 mg/L, and if intragravel oxygen levels in known spawning areas often fall below this threshold, then dissolved oxygen is believed to be a limiting factor. In another example, if stream temperatures exceed 24^{o}C for multiple and/or extended periods during summer, temperature is believed to be a limiting factor because 24^{o}C is commonly accepted as a lethal environmental threshold for most salmonid fishes (Bjornn and Reiser 1991).

In reality, critical thresholds have been established for relatively few environmental parameters, and most of these are factors that directly affect survival. Most lethal thresholds have been established experimentally in the laboratory by varying the parameter of concern while holding other aspects of the environment constant, and determining the level that kills 50% of a batch of test fish in 96 hours. While this is a very precise way of establishing lethal thresholds, results from laboratory tolerance tests may not translate directly to survival in natural streams. Fishes have evolved various mechanisms of surviving temporarily adverse conditions, and there are case studies in the literature of salmon surviving, and even thriving, in circumstances where laboratory test results predict they would perish. Bisson et al. (1988), for example, documented juvenile coho salmon exhibiting high levels of production in streams draining post-eruptive Mount St. Helens during an unusually hot summer in which water temperature reached 29.5^{o}C. Berman and Quinn (1991) found that adult chinook salmon sought pools with cool groundwater in order to maintain lower body temperatures while migrating through the relatively warm Yakima River. This is not to assert that high temperatures, elevated sediment, or depressed oxygen do not limit survival, but rather to point out that performance of fish in the field may differ from their performance in the laboratory.

The distinction between factors that may limit survival and factors that may limit individual growth rate is often blurred in practice. Most limiting factor assessments attempt to identify environmental features whose levels exceed critical survival thresholds, but there are always a number of other factors that can influence individual growth or population abundance without actually killing fish. These include factors that affect food resources and the amount of suitable rearing space. For example, Reeves et al. (1989) developed a model of assessing the factors limiting coho salmon in western Washington and Oregon. When they applied their model to data sets from Oregon coastal watersheds, Reeves et al. (1989) predicted that the amount of protected winter habitat was likely to be one of the most influential variables for this species.

Food availability is another factor likely to limit fish production in most locations without being directly lethal, but estimating the abundance of food resources in the field is very difficult. Nevertheless, we note that field experiments involving trophic manipulations in the Pacific Northwest (e.g., nutrient additions or salmon carcass introductions) have repeatedly been shown to boost the abundance of potential prey organisms, and consequently salmonid growth (Warren et al. 1964; Gregory 1980; Shortreed et al. 1984; Slaney et al. 1986; Bilby et al. 1998) while many attempts to restore physical habitats (e.g., creating pools or adding cover structures) have not produced unambiguous evidence that the restoration yielded increased populations (House and Boehne 1985; Frissell and Nawa 1992; Hilborn and Winton 1993; Reeves et al. 1997; Ward 2000). We do not conclude from these studies that food availability is somehow more limiting to salmonid populations than the quality of the physical environment. The relative importance of food and space will depend on site-specific circumstances (Chapman 1966). Rather, we believe that food has been demonstrated in field studies to be an important limiting factor, and this aspect of the aquatic environment is often overlooked in limiting factor analyses because there are no easily measured indices of food abundance.

The ISAB feels that the emphasis of a limiting factor analysis should be on whether a watershed's aquatic and riparian ecosystem processes are functionally impaired, as opposed to whether an environmental assessment reveals potentially dangerous conditions for a species of interest *at the reach scale*. Simply going to the field and observing high sediment concentrations in spawning areas or measuring high stream temperatures during mid-summer sidesteps the real issues of evaluating whether (1) the conditions observed may be perfectly normal for the site, irrespective of their suitability for target species, (2) watershed processes, e.g., inputs and routing of water, sediment, nutrients, and organic material have been significantly disrupted by human activities, and (3) local fish populations have evolved adaptations that allow them to persist in what would be considered unfavorable conditions. Instead, the ISAB recommends that limiting factor analysis include an assessment of how well the stream system is ecologically connected to its watershed, how the stream has responded to natural and anthropogenic disturbances in the past, how current and potential future conditions are constrained by land and water use, and how fish respond to the current range of conditions. We believe an analysis centered more on an examination of ecosystem processes (e.g., erosion, flow regime, aquatic and riparian interactions, large wood recruitment, and storage and routing of sediment and organic material) will produce a more meaningful picture of the conditions likely to influence the productivity of fish communities than an assessment of individual reach factors and whether they exceed putative risk thresholds.

What is the basis for predicting that correcting limiting factors will lead to measurable increases in the productive capacity of fish populations in a stream?

Many studies have shown that habitat restoration projects result in local (i.e., reach-specific) increases in fish density, but very few have demonstrated sustained response to habitat improvement at the scale of a breeding population, such as what might occupy a 3rd- or 4th-order tributary. Does this mean there is little basis for the assumption that “if you build it, they will come?” Not necessarily, because three factors regularly confound the interpretation of habitat restoration monitoring.

The first factor is the high level of natural variation that exists within breeding populations from year to year. For stream-dwelling anadromous salmonids, interannual variability (as expressed by the coefficient of variation of abundance, CV) is typically 50% or more for small populations (National Research Council 1996). Resident salmonids tend to have somewhat lower levels of natural variability, but even with a conservative CV estimate of 25% annually, almost a decade of pre-and post-monitoring population data would be needed to detect a 50% population increase with 80% certainty (Bisson et al. 1997). Since such an increase would be very difficult to achieve with habitat improvement alone, that is, with no corresponding improvement in the other Hs (hatcheries, harvest, hydroelectric operations), statistical detection of a population-level response to habitat restoration will likely take several decades at best (Hilborn and Winton 1993; Botkin et al. 2000; Ham and Pearsons 2000). That kind of commitment to long-term stream salmonid population monitoring is necessary for evaluation of the effects of habitat improvement, but has occurred at only a few locations in North America.

The second problem commonly confounding the documentation of habitat restoration success is that the scale of restoration projects rarely matches the geographical distribution of the fish population that is meant to receive the benefits. Restoration is typically targeted at improving habitat in a stream reach that has been significantly damaged, but rarely do restoration projects affect more than a small fraction of the overall breeding or rearing area. One of the few watersheds in which comprehensive, population-wide habitat restoration has been attempted is

Fish Creek, a tributary of the Clackamas River (a Lower Columbia River watershed in Oregon). In this drainage, over 1,000 instream structures were placed in Fish Creek and its main tributaries, an off-channel pond was created for winter habitat, and road crossings that blocked or impeded fish passage were either made fully passable or obliterated. After about 15 years of intensive effort, the effects of restoration on steelhead and coho salmon are not completely clear (Reeves et al. 1997). Likewise, almost two decades of restoration of Vancouver Island's Keogh River have proved insufficient to enable the effects of watershed-scale habitat improvement on steelhead to be differentiated from the effects of climate change (Ward 2000). These case studies emphasize the importance of long-term monitoring to evaluate the effects of habitat improvements.

Finally, the experimental design of most monitoring efforts to date has been deficient. Rarely are appropriate reference sites associated with treatment locations. One of the key factors complicating the interpretation of fish response to the watershed-level experiments conducted at Fish Creek (Reeves et al. 1997) and the Keogh River (Ward 2000) was the lack of an untreated, reference watershed. The problems with interannual variation in fish abundance caused by factors other than condition of freshwater habitat may be significantly reduced by the inclusion of reference sites. Most monitoring efforts also fail to consider response variables other than fish abundance. Variables such as smolts produced per spawning female, distribution of fish within the watershed, life history stage specific survival rates and growth rates all may be less variable and more sensitive to certain restoration approaches than fish density. However, these types of measures have rarely been used in assessing restoration efforts.

The implication of this pessimistic assessment of the evidence for habitat restoration effectiveness is that we still have much to learn about the efficacy of different types of restoration projects. Additional intensively monitored *population-scale* restoration efforts are needed, as most existing long-term studies have taken place in coastal, rainforest-dominated watersheds, the results of which may not be directly applicable to interior semi-arid drainage systems. One way to carry this out is to establish a regional network of experimental watersheds (approximately HUC-6 in size) in which habitat improvement can be implemented at a scale that includes most of the freshwater life history stages of a population, and to apply restoration treatments in such a way that population changes can be distinguished from other natural and anthropogenic sources of environmental variation (e.g., Walters et al. 1988, 1989). Such an effort will require a high level of organizational cooperation and coordination.


A number of analytical tools have been used to guide implementation of restoration projects in small streams and large rivers. These tools attempt to relate habitat to fish abundance in a quantitative way so the outcome of restoration projects can be predicted.

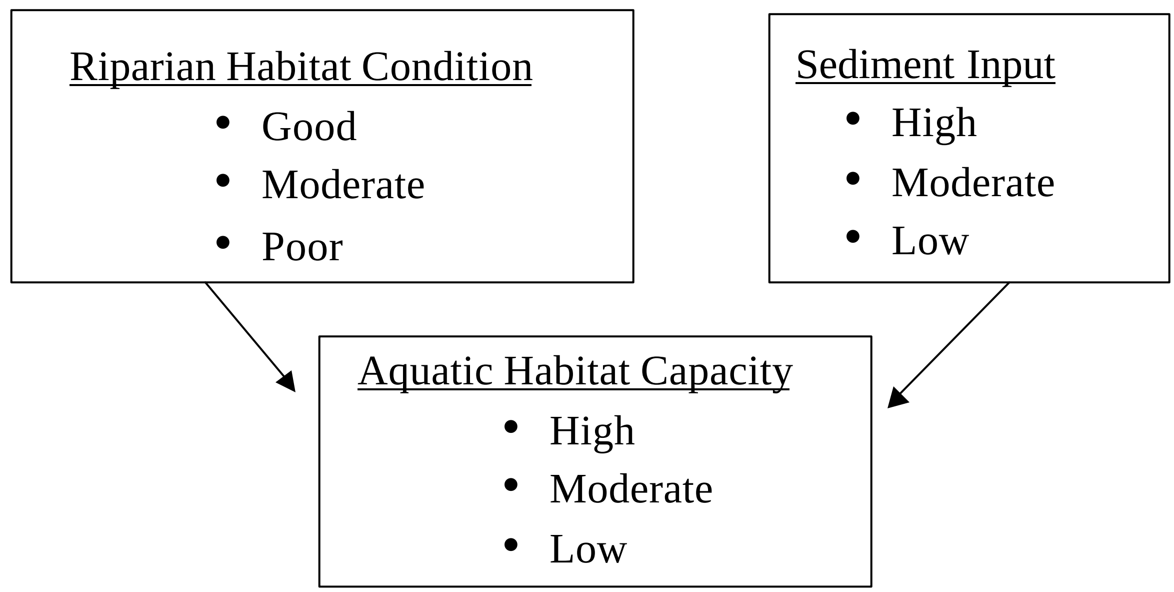
III. Implementation Strategies

The purpose of a subbasin assessment is to provide the information necessary to underpin the habitat restoration strategy that is ultimately articulated in the subbasin plan. The assessment should include three key elements: the locations where restoration efforts are apt to be most effective, the types of restoration actions that are appropriate at a given location, and some indication of the expected fish response. With sufficient data on habitat condition and sensitivities to human actions and fish populations, there are numerous analytical methods that can be used to develop this information. However, data are never complete and the menu of available analytical tools has not yet been subjected to adequate field evaluations. Predictive methods have been utilized for relating tributary habitat to aquatic community condition or the abundance of key species when less than complete data are available. These techniques are of three basic types: expert opinion, expert systems, and empirical models.

Expert opinion has been the method most commonly applied in the Columbia Basin in the development of habitat restoration projects prior to the implementation of the subbasin assessment and planning process. The expert opinion method involves compiling the opinions of local fish managers and scientists (i.e., knowledgeable experts) with regard to the sites within a subbasin that have the greatest restoration potential and the types of restoration actions that are suited to those locations. This approach is highly subjective and the assumptions underlying the opinions are often not field verified, or even made explicit. As a result, the products of such an approach are generally less helpful than those utilizing more formal decision processes, especially those with quantitative information.

Expert systems represent a formalized method of organizing and applying information and opinion. Most expert systems utilize quantitative information when available, but usually rely primarily on expert opinion. These approaches enjoy an advantage over more informal methods of using expert opinion in that they attempt to make the assumptions underlying an opinion explicit and clear. In the Columbia River Basin, two expert systems have been used extensively to assess the response of fish populations to the application of various land management practices or restoration efforts: the Interior Columbia Basin Ecosystem Management Plan (ICBEMP) Bayesian belief network (Quigley and Arbelbeide 1997) and the Ecosystem Diagnosis and Treatment (EDT) method (Mobrand Biometrics 1999) endorsed by the Northwest Power Planning Council.

The ICEBMP Bayesian Belief Network (BBN) assessment was based on a fairly simple “box and arrow” conceptual model of the effects of various land use practices on aquatic habitat condition and fish population status (Figure 4). The boxes, or nodes, of the model represented various ecosystem attributes, such as rate of sediment delivery or riparian condition. Nodes were connected to other nodes they impacted or that directly impacted them. Thus, sediment input and riparian condition were two of the nodes determining habitat capacity for salmon. In turn, factors such as grazing intensity or the frequency of fires would affect riparian condition, and road density and soil erosion would influence sediment input. Relationships between the nodes were quantified using empirical information, when available; however, in many cases data were not available. Therefore, causal connections among most of the nodes were inferred using opinion gathered from a panel of scientists with the appropriate expertise. The process of applying the expert opinion to the Bayesian Belief Network was highly formalized. Each node was assigned several possible states, usually expressed qualitatively. The conditional probability of the state of a node, given differing combinations of states of the input nodes, was estimated by each panel member and the average used to parameterize that set of conditions in the network (Figure 4). All node interactions throughout the model were parameterized in this fashion. The model was run for each of the 6^{th}-code HUCs within the interior Columbia Basin for a number of species of salmon and trout to evaluate the response of aquatic habitats and fish populations to the application of several land management alternatives.



Sediment Input	Riparian Condition	Habitat Capacity		
		High	Moderate	Low
High	Good	40	20	20
High	Moderate	20	40	40
High	Poor	5	15	80
Moderate	Good	75	20	5
Moderate	Moderate	40	50	10
Moderate	Poor	20	50	30
Low	Good	90	8	2
Low	Moderate	75	20	5
Low	Poor	30	50	20

Figure 4. Theoretical example of the process used to parameterize the ICEBMP Bayesian Belief Network. The interaction between riparian condition and sediment input can result in high, moderate or low habitat capacity. The probability (%) that a specific combination of the two input variables will produce a certain habitat capacity state is shown in the table. These probability values are the average of the probabilities assigned by each member of the expert panel that participated in the parameterization process.

Ecosystem Diagnosis and Treatment is an expert system that has been widely used in the Columbia Basin, as well as elsewhere in the Pacific Northwest. Like the ICBEMP Bayesian network, EDT is based on a conceptual model of habitat condition and fish population response. However, the EDT model is far more complicated (i.e., it contains far more boxes and arrows) than the BBN system used in the ICBEMP analysis. EDT is designed to utilize fine-scale data on habitat condition. In the model for the entire Columbia Basin the habitat data (where available) were compiled for each subwatershed. In many watershed-scale applications of EDT, habitat data for each kilometer of stream reach are required. The biological response data required for EDT also are very detailed, including life stage-specific carrying capacity and survival values and how these are influenced by habitat change. The survival and capacity values for each life history stage in each stream reach are summed through the freshwater residency of the fish to estimate smolt production, and in the process develop a "survival landscape". The EDT model also can include marine life stages of the fish, thereby enabling the projection of returning adults and population growth rates. For most subbasins, only a small proportion of the locally derived data required to parameterize EDT is available. Therefore, population capacity and survival values, as well as some of the habitat attributes, have been obtained through expert opinion. Because the fine-scaled nature of EDT requires a very large number of values to be derived by expert opinion, many of the assumptions made during the process of parameterizing the model are often not fully documented. Neither the ICBEMP-BBN nor EDT has been evaluated in a side-by-side comparison of model accuracy based on actual data.

The two expert system approaches represent an improvement over undisciplined application of expert opinion. They provide a mechanism for objectively, and in some cases transparently, combining opinions from multiple scientists. They can also provide a clear indication of the assumptions underlying the model predictions, if the process is properly documented. Parameterization of these models can provide a very good indication of key uncertainties in the system, thus highlighting areas where the collection of data might be most useful. However, their outputs must be interpreted with a full appreciation of the limitations imposed by the model structure and underlying assumptions, because both models have yet to be locally calibrated. Limitations in model structure can prevent the inclusion of key variables. For example, neither of the two expert systems addresses temporal population variability. Population response to factors such as variation in climate or periodic catastrophic events cannot be modeled, and it is becoming increasingly apparent that the productivity and diversity of stream-dwelling fishes is significantly influenced by temporal changes in habitat characteristics (Reeves et al. 1995). Nonetheless, these tools can provide information useful in developing a restoration plan.

A third approach to assessing habitat-fish population relationships is empirical modeling, which for the most part is based on local data. Although these techniques are most informative when comprehensive data on fish populations and habitat conditions are available, they can be used even when data are incomplete. The ability to use empirical approaches has been enhanced by new statistical approaches and a steady improvement in remote sensing and spatial mapping technologies. Generally, empirical modeling relates the abundance or presence-absence of the species of interest to habitat attributes at a given location (Manly et al. 2002). From this comparison, associations between populations and habitat characteristics are developed and can then be used to predict the distribution and abundance of a species or a species assemblage for locations where only habitat information is available. The predicted distribution can be used to direct protection and restoration actions to locations with the greatest potential (i.e., the appropriate suite of habitat conditions) to support target species.

These types of analyses have been used in the Columbia Basin, and elsewhere, for many years. Recently, the application of new analytical approaches coupled with information generated by remote sensing and GIS has enabled the examination of these relationships at larger, more fish-relevant, spatial scales. A classification regression tree method (CART) has been used to identify relative strength of salmonid populations at a 6th-code HUC level across the entire interior Columbia Basin (Rieman et al. 1997; Thurow et al. 1997). In this application, the CART technique was applied to several thousand HUC 6 watersheds. The method evaluated habitat variables one at a time to identify the single variable most closely associated with fish abundance for the entire data set. This habitat variable was then used to segregate the HUC 6 watersheds into two condition classes and the process was repeated for each set of sites. The sites were segregated into an increasing number of classes, each associated with a habitat attribute that was responsible for that classification assignment. Ultimately, a broad-scale picture of the habitat attributes most closely associated with abundance of different species was generated.

The NOAA Fisheries Watershed Research Program has used the technique of hierarchical linear modeling to determine habitat–abundance associations for spring chinook salmon or steelhead in the Salmon River, Willamette River Yakima River, and Wenatchee River watersheds (Feist et al. *in review*; Steel et al. *in prep*.). This approach has been called the Salmon Watershed Assessment Method (SWAM). The method regresses habitat variables compiled at the reach or watershed level, and generated primarily from remotely sensed or geospatially-referenced data, against normalized fish population abundance values for each year of record. Habitat variables that consistently exhibit a significant association with abundance are identified. The variables exhibiting a significant association with fish abundance are used in a predictive model of capacity, and the model can then be applied to areas for which fish data are not available. Model outputs enable the identification of locations within a subbasin possessing habitat conditions needed to support large populations of the target species. Both CART and SWAM techniques provide habitat characterization of sites that support high fish densities. With this information, the potential of all tributaries within a subbasin to support species of interest can be predicted from habitat attributes. Similarly, logistic regression methods can be used to identify habitat conditions associated with the presence-absence of a particular life history stage of a species (Manly et al. 2002).

The ability to use empirical assessment approaches, and the value of the output, is largely dependent upon the availability of appropriate local data. The most useful information is generated from fish population measures taken over a long period of time at multiple locations in a watershed, as well as comprehensive data on habitat condition and how that condition has changed over the period for which fish data are available. Unfortunately, these data are available for relatively few sites and must be interpreted with some caution. Estimates of fish abundance at multiple locations are largely restricted to spawner or redd counts. This life history stage may not be the most sensitive to freshwater habitat condition. Information on climate, land use patterns, and even channel characteristics, are available at many locations. But nutrient or riparian data may be much more difficult to obtain. Incomplete data may lead to erroneous conclusions about the relative importance of different habitat factors.

Until the accuracy of various decision support approaches is properly evaluated on the ground, the ISAB believes the best approach to the development of habitat recovery plans is to use more than one analytical method. Policy makers may be helped to gauge the degree of scientific uncertainty by comparing the output from two or more analytical tools, understanding the assumptions made by the analytical tools, and knowing the areas of agreement and disagreement among them. If multiple analytical techniques based on different assumptions and methods all point to the same locations and habitat conditions as being important factors limiting fish production, considerably more confidence can be placed in these conclusions. Areas where considerable divergence in model output occurs also provide worthwhile information. Attempting to understand why there are differences in model output can help identify areas where our current understanding of system function is incomplete and highlight those issues for which additional data are needed. We emphasize, however, that identification of useful analytical tools will require carefully designed comparisons of model outputs with actual habitat and fish population data.

Application of multiple analytical procedures in the stressful context of subbasin assessment and planning may seem onerous to those required to complete these processes. Nonetheless, use of multiple analytical approaches is one of the best methods of identifying key uncertainties in our understanding of the interaction between watershed condition and salmon recovery and will help ensure that planned restoration actions address habitat conditions that are controlling fish production. The difficulties in applying multiple analytical tools is somewhat alleviated by the fact that some of these analyses have already been completed for many subbasins and watersheds within the Columbia basin. The CART analyses and Bayesian Belief Network described above have been applied to all 6th Code subwatersheds in the Columbia Basin east of the Cascade Mountains. Numerous EDT evaluations have been completed. SWAM analyses have been conducted for the Salmon River, John Day River, Wenatchee River and Willamette River. Information from these completed analyses should be utilized in conjunction with output from modeling efforts undertaken during the process of subbasin assessment to provide the technical foundation for subbasin plans. These analyses can be continuously improved in subsequent subbasin assessments as new data become available and as improved analytical tools and models are developed.

IV. Implementation Incentives

Once potential restoration locations and implementation strategies have been identified and prioritized, the question becomes how to ensure that strategies are implemented and restoration objectives are achieved. The successful implementation of tributary habitat improvements rests in large part on the incentives facing those with decision authority over the resources of concern.

In the case of publicly owned resources – either state or federal – mechanisms exist through various pieces of legislation, administrative rules and consultation procedures to motivate and coordinate restoration actions. Although problems of inter-jurisdictional coordination do exist, they are mitigated by the fact that the incentives facing those making decisions over public land tend to be aligned with the public objectives of tributary habitat restoration.

In contrast, when tributary resources are privately owned, no mechanism exists to naturally bring together the public demand for habitat improvements with the private supply of restoration actions. For goods and services exchanged through markets, the market aligns demand and supply through the responsiveness of price to changing conditions. But many of the environmental goods and services involved in tributary habitat – water quality, riparian vegetation, biodiversity- are amenities that remain outside a market. Something other than price has to encourage people to take certain actions so that private supply of environmental goods and services matches public demand.

a. Environmental policy instruments

The policy instruments used in the United States to encourage people to do the “right thing” for environmental resources fall into three general categories of motivation: regulatory coercion, economic incentives and moral suasion.

Regulatory coercion is the motivation found in state and federal regulatory programs that are based in enabling legislation and enforced through government authority. Compliance is a matter of law. Examples are water temperature standards and pollution TMDLs established under the Clean Water Act and land use controls established under the Endangered Species Act.

Economic incentives use self-interest as the motivation for action. Individuals are penalized for undesired actions or rewarded for desired actions, and have flexibility to select the type and magnitude of participation. Examples are charges for point-source pollution discharges (negative incentives) and payments for agricultural acreage set-asides (positive incentives).

Moral suasion uses the motivation of “doing the right thing” for the environment. A positive feeling of contributing to the public good underlies much volunteer work and voluntary actions. Examples are volunteer adopt-a-river programs and voluntary watershed councils.

The type of policy instrument most appropriate for tributary habitat restoration depends on the context of the restoration. In general, restoration strategies will be multifaceted and will include resources regulated under mandatory programs as well as those without specific protections. To the extent that the resources are privately owned, strategies must go beyond regulatory coercion to motivate people on the basis of either suasion or self-interest.

Voluntary programs are an increasing phenomenon in environmental restoration. The system of Oregon watershed councils is based on voluntary participation, as are many stream restoration projects and activities. Voluntary programs have the advantage of being low cost and flexible in testing new approaches, but they have the disadvantage of uncertain effectiveness (EPA 2001).

Incentive-based policy instruments can provide either positive incentives to encourage desired behavior or negative incentives to discourage environmentally harmful activities. Cost-share and incentive payment policies are frequently used as positive incentives to encourage changes in agricultural practices that will protect environmental resources or to take land out of agricultural production (USDA 2000). Some involve the government taking on partial legal interests in the land through conservation easements (Wiebe et al 1996). Pollution emission fees are negative incentives designed to increase the costs of undesired behavior. Compliance mechanisms like the

Wetland Conservation ("Swampbuster") Program are a mix of positive and negative incentives, in which a basic level of environmental compliance is a condition of eligibility for other programs (USDA 2002a).

The U.S. experience with environmental restoration has shown a number of benefits from applying the motivation of self-interest to induce environmentally desirable private behavior (Shogren and Tschirhart 2001). The experience most closely related to tributary habitat restoration is in the suite of federal incentive-based agriculture/environmental programs administered by the USDA. These programs use positive economic incentives to create changes in agriculture that will promote environmental goals. They operate on the basis of compensating producers for the cost of lost production opportunity brought by changing uses of agricultural land. Most would apply to tributary habitat restoration where riparian land is in agricultural production.

b. Federal agriculture-environmental incentive-based programs

The 2002 Farm Act continues the trend of increasing funding for agricultural programs targeted at achieving environmental benefits. In contrast to past conservation funding which emphasized the retirement of land from agricultural production, emphasis has shifted toward conservation actions on working land. Land retirement programs place a stronger emphasis on wetland restoration (USDA 2002b). The following are some examples of federal incentive-based agricultural programs that would apply to tributary habitat restoration. These programs would be compatible with the "range of natural conditions" management objectives.

Conservation Reserve Program (CRP): provides farm operators an annual per-acre rental payment and cost-sharing for establishing a permanent land cover, in exchange for retiring environmentally sensitive cropland from production for 10- to 15-years. Offered land is evaluated on the basis of an Environmental Benefits Index (EBI).

Wetlands Reserve Program (WRP): provides cost sharing and/or long-term or permanent easements for restoration of wetland on private land. The 2002 Farm Bill increased the acreage cap 1.2 million acres (117%). The landowner voluntarily limits future use of the land, yet retains private ownership.

Environmental Quality Incentives Program (EQIP): provides technical assistance, cost-sharing and incentive payments to assist livestock and crop producers with conservation and environmental improvements through 5-10 year contracts. It is targeted to watersheds, regions, or areas of special environmental sensitivity identified as priority areas.

Wildlife Habitat Incentives Program (WHIP): provides cost-sharing assistance to landowners for developing habitat for upland wildlife, wetland wildlife, threatened and endangered species, fish, and other types of wildlife. WHIP funds are distributed to states based on their wildlife habitat priorities, which may include wildlife habitat areas, targeted species and their habitats, and specific practices.

Conservation Security Program: this new program provides payments to producers for maintaining or adopting a wide range of structural and/or land management practices that address a variety of local and/or national resource concerns.

Small Watershed Program: provides technical and financial assistance to states, local units of government, tribes, and other sponsoring organizations to voluntarily plan and install watershed-based projects on private lands for watershed protection, fish and wildlife habitat enhancement, and wetland restoration, among other purposes.

c. Federal programs

Challenge Cost-Share Programs (CCS): The U.S. Forest Service provides funds and technical expertise for cooperative projects to improve aquatic habitat, fishing opportunities, and environmental education. The U.S. Fish and Wildlife Service also funds challenge cost-share programs targeted at conserving fish and wildlife resources and natural habitats on public and private lands. CCS funds must be matched by contributions of money, labor, equipment, or materials from conservation groups, private enterprises, individuals, schools, or other public agencies (USDA 2002c; USDI 2002).

d. Programs for environmental improvements

Some state actions for environmental restoration are the implementation of federal regulatory programs. For example, the Environmental Protection Agency delegates authority to the Oregon Department of Environmental Quality (DEQ) to conduct federal programs to implement the Clear Air Act, the Clean Water Act and the Resource Conservation and Recovery Act (control of hazardous waste). To implement the Clear Water Act, the DEQ maintains the 303(d) List of stream segments that do not meet the established water quality standards, established to protect native salmon and trout. In addition, the DEQ develops TMDLs (Total Maximum Daily Loads) for a range of pollutants delivered through both point and non-point sources. Management plans to restore streams and rivers bodies to water quality standards are then developed in cooperation with landowners (DEQ 2002). These programs are targeted to achieving fixed standards mandated by federal law.

Other state programs for habitat recovery operate on a voluntary basis, but without the large component of economic incentives contained in federal programs. An example is the Oregon Watershed Enhancement Board (OWEB), which provides grants for watershed council support, watershed assessment and monitoring, watershed action plan development, watershed restoration project design and implementation, and watershed education and outreach projects. A funding directory for Watershed Enhancement is contained on the OWEB website (www.oweb.state.or.us/directory/fundingintro.html).

e. Effectiveness of incentive-based programs

The growth in voluntary compliance with incentive-based programs in agriculture indicates that they have been popular with agricultural producers. They have also been effective in diminishing agricultural impacts on environmental resources. Incentive-based tools take account of basic principles of economic behavior in ways that tend to lead to more effective environmental protection. A disadvantage of the agriculturally based incentive programs is that they do not apply to all crops, to parcels of land below a minimum size, or to land outside of agricultural production. In areas of rapid population growth and urbanization, similar incentive-based programs are needed to encourage nonagricultural participation in tributary habitat restoration. The principle of using compensation to provide an economic incentive for restoration actions applies across sectors and scale of ownership.

V. Foundation for Habitat Recovery

Recent studies have shown that there are significant inherent differences in the productivity of salmon streams. Research at the Northwest Fisheries Science Center suggests that most of the salmon production for a subbasin originates from a relatively small number of watersheds or mainstem reaches (Pess et al. 2002; Feist et al. *in review*). Geology, elevation, topography, surface and subsurface hydrology, and vegetation all seem to be influential. Likewise, land and water uses affect productivity. While there have been many investigations of the influence of local and reach-scale environmental factors on habitat suitability for salmonids, the importance of landscape-scale features has received far less attention. Furthermore, stream productivity and biological diversity is known to vary with climate cycles and large natural disturbances. Providing productive conditions for all species of interest may depend upon watersheds exhibiting a variety of habitat conditions produced by variations in the underlying physical attributes of stream reaches and the process of disturbance and recovery. The unavoidable conclusion is that salmonid productivity varies widely in both space and time, and it will never be possible, or desirable, to maintain all streams in a condition considered optimum for one species of fish.

Understanding large-scale spatial and temporal patterns is important in subbasin planning because restoration strategies need to identify the best sites for protection as well as the best candidates for restoration – those sites that have a likelihood of becoming highly productive at some future time. Subbasin planners will not have the resources to do everything everywhere; difficult decisions will be needed to apply limited resources to the most cost-effective restoration projects. In preparing this report, the ISAB heard from scientists who have completed subbasin assessments that have been considered successful. These assessments included analyses of the Skagit River (Washington) Basin, Oregon coastal watersheds, and the Willamette River Basin. Although the objectives of the assessments varied, the approaches that contributed to their successful completion shared four common themes.

First, the assessments began with a *thorough, systematic inventory of conditions within the drainage system*. Instead of gathering as much data as possible and then attempting to make sense of it all, the investigators first identified the information needed to address their questions and then set up data collection programs to acquire it. In some cases, the information had already been obtained by other organizations, so data compilation became a matter of putting it into the proper format for analysis.

Second, successful assessments utilized *measures of ecosystem functionality that extended beyond the stream reach scale to encompass whole watersheds*. In each situation, rule sets were established that enabled investigators to make statements about the general ecological condition or "health" of an area based on combinations of environmental factors. This step was critical because it provided the context needed to assess the significance of habitat degradation at individual sites, as well as a means of judging the potential for effective restoration at different locations. The rules (decision criteria) for linking watershed condition with ecological health and biological productivity may have been derived from the knowledge of local experts, scientific literature, or correlations between landscape features and fish abundance, but in all cases these rules were made explicit in the analytical process. This was important to understanding and interpreting results.

Third, the distribution and habitat requirements of target species over the entire freshwater life cycle were overlain on spatially referenced maps of watershed condition to allow for a *landscape-scale comparison of life-history needs and habitat status*. This step required that the seasonal distribution of fishes be known or inferred from reasonable models, a step that usually required field verification. Superimposing fish population data on landscape condition allowed investigators to identify locations within subbasins where both fish populations and ecosystem processes were healthy, fish populations or habitat conditions were depressed, or where both populations and habitats were impaired. Streams having strong populations and intact ecosystems became candidate "stronghold" watersheds that could serve as sources from which wild fish populations could be rebuilt in restored tributaries. Locations having weak populations but relatively high quality habitats could be made more productive by providing more spawning fish through changes in harvest management or improved access to these sites. Locations where populations were relatively (or potentially) abundant, but habitats were impaired, could become excellent candidates for restoration. Tributaries where populations were weak and habitat was degraded were recognized as locations in which restoration would be expensive and risky, unless both fish population access and sustainable habitat quality issues were addressed.

Fourth, successful subbasin assessments all contained *explicit strategies for restoration* (e.g., rebuilding outward from core stronghold areas; reconnecting aquatic, riparian, and floodplain systems) and provided a means of forecasting future conditions based on projected human developments in the area of interest. The best example was the Willamette River Basin assessment (www.orst.edu/dept/press/WillRivrBas.html), in which the influence of alternative land development patterns on the distribution and abundance of high quality aquatic habitats was modeled. The Willamette River Basin analysis (Hulse et al. 2002) utilized different combinations of active and passive restoration techniques in its visual projections of future habitat scenarios. It serves as a model of how a thoughtful subbasin assessment can be used to effectively inform restoration decision-making in the Columbia River Basin.

Tracking the effects of habitat improvement projects is critical to judging their success, but monitoring and evaluation can be time-consuming and costly. How can monitoring and evaluation be organized to make the most of scarce resources?

VI. Monitoring and Evaluation

Understanding the effect of habitat conditions on salmon population performance requires replicated observational studies or intensive research level experiments to be conducted at large spatial and long temporal scales. Few evaluations of tributary habitat in the Columbia River Basin meet these criteria.

Monitoring has been categorized in a hierarchical sequence (Tier 1, Tier 2, or Tier 3) in the NMFS All-H document (Conservation of Columbia Basin Fish: Final Basinwide Salmon Recovery Strategy, Volume 1, Table 4) and repeated in the 2000 BiOp. For consistency, we discuss monitoring and make our recommendations in terms of this hierarchical sequence. The three levels are: trend monitoring (Tier 1), statistical monitoring (Tier 2), and research monitoring (Tier 3).

Tier 1 (trend or routine) monitoring obtains repeated measurements, usually representing a single spatial unit over a period of time, with a view to quantifying changes over time. Changes must be distinguished from background noise. For example, the temperature of water entering and leaving a habitat improvement site might be measured in August every third year for a 21-year period. Tier 1 monitoring can be applied to individual project sites or to a large area. For applications involving large areas, the ISAB anticipates that remotely sensed, spatially referenced data would be used. For example, long-term trend monitoring of changes in riparian and other terrestrial habitat could be accomplished with aerial photography or LandSat imagery. In general, Tier 1 monitoring does not establish cause and effect relationships and does not provide statistically inductive inferences to larger areas or time periods. However, Tier 1 trend monitoring on similar projects replicated over time and space can provide compelling evidence for general conclusions. Also, aerial photography or data layers in a GIS yields a census of the study area thus eliminating the need for spatial sampling and statistical analysis at the scale studied. However, many habitat attributes cannot be derived from remotely sensed data and measurement errors may limit the usefulness of remote sensing for detecting changes.

Tier 2 (statistical) monitoring provides statistical inferences to parameters in a study area as measured by certain data collection protocols. These inferences extend to larger areas and longer time periods than the sample. The inferences require both probabilistic selection of study sites and repeated visits over time. A good example is the process used to estimate abundance of spawning coho salmon coastal Oregon watersheds, described in the Oregon Plan for Salmon and Watersheds Monitoring Program (www.nwr.noaa.gov/pcsrf/Moore/). This method utilizes a rigorous design for probabilistic site selection to ensure that the locations where spawning coho salmon are enumerated are representative of all stream reaches. Evaluation of the effectiveness of habitat improvement projects will require Tier 2 statistical monitoring for some parameters, e.g., the number of redds in a watershed.

Tier 3 (experimental research) monitoring is often required to establish cause and effect relationships between management actions and population response. Bisbal (2001) defines this level of effort as *effects* or *response monitoring*; the repeated measurement of environmental variables to detect changes caused by external influences. Tier 3 research monitoring requires the use of experimental designs incorporating "treatments" and "controls" randomly assigned to study sites. Generally, the results of Tier 3 research monitoring qualify for publication in the refereed scientific literature. Examples of Tier 3 monitoring include projects to evaluate the effects of different levels of fertilization on growth and survival of juvenile salmonids with streams selected randomly for reference and treatment, and projects to evaluate the effectiveness of various land restoration or management techniques.

The value of research, monitoring and evaluation efforts is greatly enhanced if these elements are integrated. Tier 1 and Tier 2 monitoring results will help define the key issues that should be addressed with a more intensive Tier 3 effort. Results of the Tier 3 research will identify which habitat attributes are most informative for Tier 1 and 2 efforts. Implementing Tier 1 and 2 monitoring without a corresponding Tier 3 effort would not provide conclusive information about the efficacy of various restoration approaches. However, without the more extensive Tier 1 and 2 monitoring, there would a great deal of uncertainty regarding the extension of results from the geographically focused Tier 3 monitoring to sites not being studied. The actions listed below are the essential elements of a research, monitoring and evaluation plan to evaluate tributary habitat restoration.

- Develop a sound Tier I trend-monitoring procedure based on remotely sensed data obtained from sources such as aerial photography or satellite imagery. Changes in terrestrial and aquatic habitat and land use should be monitored at the largest possible spatial scale using the "finest grained" data available.

- Develop and implement a long-term statistical monitoring program (Tier 2) to evaluate the status of fish and wildlife populations and habitat. This action would entail development of probabilistic (statistical) site selection procedures and establishment of common protocols for cost-effective "on the ground" or remotely sensed data collection of a limited number of indicator variables. Measurement of indicator variables should be done on the same sites. Status Monitoring plans are being developed by the Action Agencies for implementation of the EPA EMAP probabilistic selection of aquatic sites in a pilot project in the John Day, Upper Salmon, and Wenatchee Subbasins (BPA Draft Report "Research, Monitoring & Evaluation for the NMFS 2000 FCRPS Biological Opinion"). Every effort should be made to include the same site selection protocols and data collection methods throughout the Columbia River Basin.

- Develop or improve existing empirical models for prediction of abundance or presence-absence of focal species as data are obtained in a Tier 2 status monitoring program. Potential predictor variables include not only physical habitat variables, e.g., flow, temperature, etc., but also habitat recovery actions. The empirical models can be used to evaluate the relative importance of physical factors and habitat improvements and to predict abundance or presence-absence throughout major sections of a subbasin (Manly et al. 2002).

- Implement a research monitoring (Tier 3) effort at selected locations in the Columbia Basin to establish the underlying causes for the changes in population and habitat status identified in Tiers 1 and 2 monitoring. An extensive, long term status monitoring program can identify important and unexplained trends and changes. Tier 1 trend monitoring and Tier 2 statistical monitoring provide indications of trend and change in indicator variables. But because Tiers 1 and 2 monitoring efforts rely on low-intensity data collection over large areas, the reason for certain trends and changes is not well understood. For example, the status monitoring may indicate that a major increase in juvenile fish production occurred between 2010 and 2020 that cannot be attributed to habitat changes derived from Tier 1 or 2 monitoring. Why? A population of bull trout is detected in an area where modeling results based on current knowledge indicate they should not exist. Why? Answering these questions requires a more thorough understanding of the ecological processes governing fish population responses to habitat condition than can be obtained through Tiers 1 and 2 monitoring. As understanding the "why?" is often critical in identifying habitat protection and restoration priorities, Tier 3 research monitoring should be included as part of the monitoring strategy.

There are two general approaches to Tier 3 field research to evaluate the effectiveness of tributary habitat restoration activities (i.e., to determine cause and effect relationships). The first approach is consistent with that promoted by the Action Agencies (Bonneville Power Administration (BPA), Corps of Engineers (Corps), and Bureau of Reclamation (BOR)), and NMFS in their RME Plan (see the BPA Draft Report "Research, Monitoring & Evaluation, for the NMFS 2000 FCRPS Biological Opinion"). In this approach a large number of pairs of sites (e.g., watersheds) might be identified where the primary difference between members of each pair is that one has a certain habitat improvement (e.g., stream fencing to exclude livestock) and the other does not. Future management actions would be uniformly applied to both members of a pair. Enough pairs of sites are obtained to generate acceptable power for standard statistical tests to detect important differences in the estimated indicator variable(s). Given the number of pairs involved, parameters that can be monitored by Tier 1 or 2 methods within a reasonable budget are limited (e.g., estimates of spawners entering the watersheds and smolts leaving).

The design of such a study will be similar to that used in the large scale "treatment-control" observational Idaho Supplementation Study (ISS). General conclusions from this approach may be obtained using regression-correlation type analyses provided that a sufficiently large number of treatment and control sites are included in the study. However, the ISAB cautions that maintaining the study design for a large number of replications over a long period of time is a difficult task. For example, in the ISS there have been changes in the original design that will make interpretation of the data difficult. Furthermore, this design requires study of one factor at a time (e.g., fencing). Study of the interactions between two different types of habitat improvements (e.g., fencing and placement of large wood) would double the number of required sites in a 2x2 factorial design.

The second approach is to focus intensive research monitoring in a small number of watersheds in each subbasin, an approach the state of Washington has termed Intensive Watershed Monitoring (IWM). This approach is similar to the "Top-Down Monitoring" described in the RME plan (BPA Draft Report "Research, Monitoring & Evaluation, for the NMFS 2000 FCRPS Biological Opinion"). The basic premise of IWM is that cause and effect relationships in complex systems can best be understood by concentrating monitoring and research efforts at a few locations. Closely spaced measurements in space and time are often required to develop a thorough understanding of the processes responsible for habitat or fish population response to a management action. Concentration of effort can focus sufficient resources and research expertise to understand some of the complex interactions governing system response to restoration activities.

There are obvious advantages and disadvantages to each approach. The first (e.g., ISS) attempts to draw inferences about a very large area by monitoring a large number of pairs of sites over the target region. Obviously, the inferences would be stronger if Tier 3 monitoring with random assignment of treatments and controls is used, but this requirement for cause and effect conclusions is likely not practical. Inferences are usually based on correlation-regression type analyses and confidence is gained in the conclusions as the numbers and geographical distribution of the study sites are increased. The primary disadvantages of the approach are costs, logistical difficulties in dealing with a large number of sites in a large area over a long time period, difficulty of locating suitable control (untreated) sites, and the ability to monitor only one factor at a time.

The second approach (IWM) limits inferences to a small number of sites with limited geographical coverage, but with intense study of more parameters and their relationships. Again, randomization of treatment and control to relatively large watersheds is probably not practical, but perhaps some randomization can take place on streams within the studied watersheds. Inferences concerning applicability of the conclusions to large regions are based on professional judgment. The primary disadvantages are cost, limited inductive inferences to large regions, and logistical difficulties of dealing with long-term studies.

The scientific debate between the relative advantages and disadvantages of these two philosophies will not be settled here. However, we recommend that the IWM approach be used to begin to develop a better understanding of the causes of biological response to tributary habitat actions. The IWM approach to research and monitoring has a proven history of effectiveness. Some of the earliest intensive watershed monitoring efforts were instituted by the Forest Service in the 1950s to better understand watershed responses to logging. Research at these sites expanded over time to include water chemistry and biological responses to logging. Changes in the regulatory framework of forest management practices nationwide have been based on studies conducted at experimental watersheds such as the H.J. Andrews Experimental Forest in Oregon, the Hubbard Brook Experimental Forest in New Hampshire, and the Coweeta Experimental Forest in North Carolina.

The success of these efforts led to a number of intensive watershed monitoring efforts in the Pacific Northwest to evaluate the response of salmonid fishes to forest practices. The Alsea Watershed Study, which was initiated in the early 1960s and continues today, examined the response of coho salmon and cutthroat trout to various logging methods in three small watersheds in the Oregon Coast Range. Early results from this study provided some of the impetus for the revision of laws governing forest practices in Oregon and Washington in the early 1970s. In the 1970s an ambitious watershed-level project was initiated at Carnation Creek on Vancouver Island, British Columbia that evaluated the response of salmonids to logging of an old-growth forested watershed. The results of this study led to a revision of the forestry code for British Columbia and also influenced revisions of forest practice rules in other areas of the Pacific Northwest. Research on trophic effects of salmon and trout at the Keogh River on Vancouver Island have provided a vastly improved appreciation for the importance of nutrients and trophic processes in supporting salmon productivity (Ward and Slaney 1988).

IWM is a method of achieving the level of sampling intensity necessary to determine the response of salmon to a set of management actions, but admittedly in limited numbers of areas. Evaluating biological responses is complicated, requiring an understanding of how various management actions interact to affect habitat conditions and how the ecosystem responds to these habitat changes. The response of fish is dependent on the relative availability of the habitat types that change through the period of freshwater rearing (Table 1), and the manner in which these habitat types are influenced by application of a management action. Additionally, the relative influence of each habitat type on fish survival changes from year to year due to variations in weather and flow, the abundance of fish spawning within the watershed, and other factors. For example, smolt production may be limited by spawning habitat during years when scouring flood flows occur during incubation and greatly decrease egg survival. When more benign flow conditions occur during egg incubation, smolt production may be influenced more by the availability of food during spring and summer, or by the availability of adequate winter habitat. Untangling the importance of various factors and predicting how these factors respond to land use actions or restoration efforts can only be accomplished with an intensive monitoring approach.

A before vs. after, control vs. impact (BACI) study design often is well suited to address many of questions amenable to IWM. This type of design enhances the ability to differentiate treatment responses from responses due to factors not directly affected by the treatments (e.g., weather). This approach also implies that IWM efforts must include two or more watersheds, with at least one serving as a reference site where no experimental treatments are implemented during the study. Obviously, confidence in the generality of the conclusions is enhanced with random assignment of treatment and control status to the watersheds although this may not be practical in many applications. A calibration period prior to applying treatments is required to determine how the reference and treatment watershed compare in the key response variables prior to any habitat manipulation. The length of time required to develop this baseline will vary among watersheds. However, recent comparisons of adult salmon densities among multiple sites suggests that relative abundance, i.e., concordance of abundance patterns, is fairly consistent (Pess et al. 2002; Feist et al. *in review*), suggesting that a fairly short calibration period may suffice. The calibration period for sites with records of spawner abundance and smolt output would be much shorter than for watersheds where these data have not been collected.

Treated and untreated sites can be paired at a multiple spatial scales within the IWM design, with the scale dependent on the question being addressed. In fact, reference sites for some reach-level projects could be located within the treated watershed. These reference sites would consist of streams comparable to the location where a restoration action is applied but where no habitat manipulation would occur during the period of evaluation. Questions that can be addressed at this finer scale include life-history specific biological responses or physical habitat responses to management actions. For evaluations of treatment effects at the scale of an entire watershed, a comparison with a nearby watershed that is not undergoing treatment is required. Therefore, the IWM approach does require sufficient management coordination to ensure that reference sites remain untreated through the duration of the study. This does not imply that any management activities in the reference watershed will compromise the integrity of the study. The validity of the study will be maintained as long as other management activities not directly related to the restoration projects being evaluated are similar at the reference and treated locations. For example, the effectiveness of restoration actions can be evaluated in watersheds being actively managed for commodity production provided that the type and intensity of agriculture or forest management activities in the treated and reference watersheds are comparable.

Fundamental to the IWM approach is the establishment of a set of overarching objectives that provide the context for the application of ecological restoration and to which individual projects can easily be related. As the goal of most habitat restoration efforts for salmon and trout is to improve the survival of fish through their entire period of freshwater residency, the objectives should relate to this outcome. Individual restoration projects should collectively contribute to the attainment of the objectives. To determine whether this is occurring, projects applied at the reach scale should be nested within, and clearly related to, the watershed-level objectives for habitat condition and fish populations. Such a nested hierarchy creates an interconnectedness among projects that is critical to assessing the efficacy of the restoration efforts.

Implementation of intensive watershed monitoring should begin with an assessment of the current condition of the watershed. The assessment will provide some indication of the factors that might be influencing fish production in the watershed. For example, if the watershed assessment identifies a lack of large wood in streams, a testable hypothesis could be that lack of pool habitat is limiting available rearing space. An experiment to evaluate this hypothesis might involve deliberate addition of wood to channel segments and measurement of the change in pool habitat and summer and winter rearing populations at these sites relative to populations at untreated reaches (reach-level evaluation). However, even if this analysis indicates an increase in the number of fish rearing at treated sites, it does not imply that wood addition projects will boost the overall productivity of fish populations. In order to determine whether the wood addition would actually change ecosystem capacity, rather than simply attracting fish to treated reaches that would have reared elsewhere, measures of watershed-level productivity are required.

To evaluate watershed-scale responses to restoration, the treatments (e.g., wood additions) would need to be applied at enough locations that population responses can be detected. If the initial hypothesis is correct and pool habitats do have a controlling influence on fish production in the watershed, the number of smolts produced or survival rate from egg to smolt should increase. The number of treatment sites required to detect a watershed-level response can be evaluated as wood-addition projects are successively implemented. Due to the expense and labor involved in wood additions to channels, application of treatments will occur over a period of years. Small increases in density of rearing fish at the reach level would indicate that watershed-scale responses would only be discernible when a large number of sites had been so treated. A very dramatic density response to restoration at the reach level might suggest that changes in populations could be measurable with treatment of fewer sites.

The minimum information required to evaluate watershed-level responses is numbers of spawning adult fish and smolt emigration. Counting fences or weirs at the downstream end of a watershed provide the most accurate measure of adult salmon returning to spawn (Botkin et al. 2000). However, weirs are labor intensive and provide no information about spawner distribution within the watershed. Counts of adults on spawning grounds or mark-recapture estimates of spawning fish or carcasses conducted during the time of spawning is not as accurate as counts at weirs in determining breeding population size, but does provide information on distribution. Application of Tier 2 probabilistic techniques of reach selection, and frequent, consistent surveys of each reach are required to provide accurate spawner estimates.

An improved statistical approach known as the USEPA Environmental Monitoring and Evaluation Program (EMAP) methods has been developed and implemented on the Oregon coast for coho salmon. Smolts leaving the watershed are sampled with traps, including fences or weirs that capture all emigrating smolts (although fences may become inoperable at high flows), or devices that capture a portion of the fish (scoop or screw traps). Partial sampling traps are easier to maintain and can be utilized in channels too large for weirs; however, these types of devices require frequent calibration to determine the proportion of smolts being captured. With adult and smolt data it is possible to calculate the survival rate of fish from spawning through smolting. The objective of nearly all salmon habitat restoration efforts is to increase this value. Regardless of the methods used to estimate adult salmon and smolt abundance, these measures are critical to any evaluation of fish response to tributary habitat restoration and should be included at all intensively monitored sites.

Augmenting the smolt and spawner data with information on egg survival and the distribution, abundance, and survival of juvenile salmon from emergence through smolting can enable salmon response to individual restoration projects to be estimated at the scale of whole watersheds.

Capturing fish seasonally by electrofishing, seining or trapping at multiple locations across a watershed enables an estimate of species distribution, abundance, growth rate, and age class composition. If handling fish is not permitted because of ESA considerations, one alternative is a visual survey over a broad area (Hankin and Reeves 1988). Although this method does not provide data on fish species and size that are as accurate as methods that involve capturing fish, it is rapid and permits sampling the entire stream network in 6^{th}-code HUC watersheds. A combination of the two approaches (capture and visual sampling), including a complete survey coupled with nested subsamples at selected sites where fish are captured and measured, would yield the most complete information.

Tagging salmon captured during stream sampling and subsequent recovery of tagged fish at smolt traps can provide additional information on survival of fish rearing in different parts of the watershed as well as the effectiveness of individual restoration projects. Differences in survival among reaches or habitat types indicates key mortality factors operating in the river, and aids in the identification of restoration efforts likely to have the greatest effect on salmon populations.

Collection of fish population data can be coupled with information on changing habitat conditions and climate. Salmonids are very sensitive to variations in flow, temperature, and other factors that are not directly influenced by restoration treatments, and interpretation of data can be enhanced by collection of this information. At a minimum, a recording flow gauge is required at the mouth of the reference and treatment watersheds. In addition, if some of the restoration efforts result in altered flow patterns, secondary flow gauges should be installed at locations where these efforts are undertaken. Weather stations recording precipitation and air temperature should be located near the downstream end of the watershed. Water temperature should be recorded year round at each gauging station and at all sites where the objective of a restoration action is to alter water temperature. Instruments to record flow, weather and water temperature information have improved dramatically in the last decade and costs have decreased. However, maintaining the instruments and the environmental databases are labor intensive. Thus, this type of sampling is well suited to the intensive watershed monitoring approach.

Habitat measurements should be collected concurrently with juvenile fish sampling; these data are especially important at sites where restoration projects will be implemented. Physical characteristics of the channel (e.g., pools and riffles), riparian vegetation, sediment in pools and spawning gravel, water quality (e.g., temperature, dissolved oxygen, and turbidity), and nutrient levels, are examples of habitat features that may be relevant to restoration efforts. Projects designed to increase pool habitat will focus on the physical attributes of the channel, while measures of nutrient levels, and primary and secondary production, would be appropriate measures of a salmon carcass project. The expense and effort needed to obtain the data necessary for evaluating the response of salmonids to habitat restoration is considerable, and this supports an approach of focusing intensive monitoring efforts on a relatively few locations. It is likely to require several fish generations to get statistically supported answers to questions about the effectiveness of habitat restoration. However, by implementing these evaluations with clear objectives, careful employment of experimental and statistical design, disciplined adherence to the experimental constraints in treatment and reference sites, and patience, results can be obtained that will greatly improve our ability to promote salmon recovery.

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