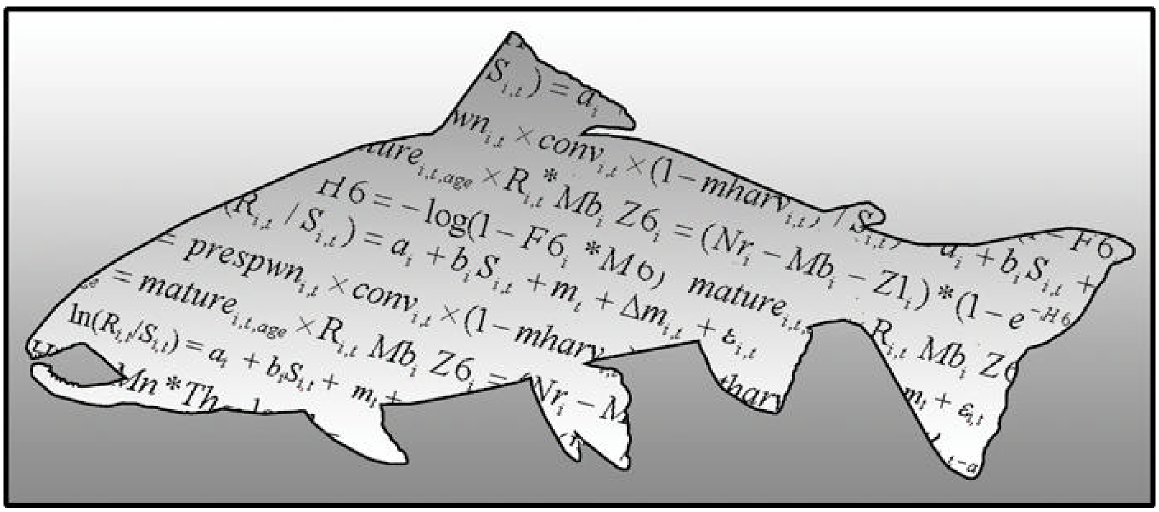
Independent Scientific Advisory Board
for the Northwest Power Planning Council
and the National Marine Fisheries Service
851 SW 6th Avenue, Suite 1100
Portland, Oregon 97204

Model Synthesis Report

An Analysis of Decision Support Tools Used in Columbia River Basin Salmon Management



March 2, 2001
ISAB 2001-1

Peter A. Bisson
Charles C. Coutant
Daniel Goodman
Robert Gramling
Dennis Lettenmaier
James Lichatowich
Eric Loudenslager
William Liss
Lyman McDonald
David Philipp
Brian Riddell

ISAB Model Synthesis Report

Contents:

ISAB Model Synthesis Report

An Analysis of Decision Support Tools Used in
Columbia River Basin Salmon Management

Introduction

Several major analytical efforts are underway to support decision-making for salmon restoration in the Columbia Basin. These efforts include NMFS' Cumulative Risk Initiative (CRI), the Plan for Analysis and Testing Hypotheses (PATH), the Northwest Power Planning Council's use of the Ecosystem Diagnosis and Treatment protocol (EDT), the US Forest Service's and Bureau of Land Management's effort undertaken for the Interior Columbia River Basin Ecosystem Management Plan (ICBEMP-BBN), and the Columbia River Intertribal Fish Commission's COHORT model (CRITFC). Within the region, conflict has developed over the role these models should play in the decision-making process. Part of this conflict is manifested in the current debate over the need for multiple modeling approaches, such as CRI and EDT, when considerable time and money has already been invested on the PATH models. This debate is exacerbated because the models were developed for different purposes and they take different analytical approaches, use somewhat different data sets, and make different assumptions.

Policy-makers (and the public) are faced with important decisions concerning salmon restoration. The conclusions of the various models contribute to the decision-making process. However, because the objectives, data, and assumptions vary among the models, it is not surprising that their conclusions differ. Decision-making could be hampered if the region becomes engulfed in a "collision of models," with various interest groups and agencies advocating the conclusions that best support their interests and mandates. Such a debate will undoubtedly occur as an inevitable consequence of the gravity and implications of impending decisions, and the lack of creditable scientific conclusions concerning the probability (feasibility) of outcomes following any particular management intervention. The ISAB undertook this project with the intent of bringing some clarity to the regional debate. The purpose of the project was two-fold:

1. *Comparative Synthesis* - - to clarify the questions or problems that each model was designed to address and to provide an overview of each of the models and a synthesis that describes both consensus conclusions and areas of disagreement among the models. We focused on the main results or conclusions of the models and did not perform an in-depth evaluation of the structures of the models and the quality of data used to calibrate and validate them. Given the complexities of the models and the histories of their development, the latter would be a daunting task.

2. *Assessment of the Role of Models in the Decision-making Process* - - to assess how science interfaces with decision-making, specifically relating to the roles models play in the process.

In undertaking this project, the ISAB requested the assistance of the principal scientists involved in each analytical approach. We asked that they 1) provide written responses to a series of questions about their modeling approach (the *verbatim* responses are attached in the Appendix III), 2) give an oral presentation of their responses and participate in a discussion of the responses with the ISAB, and 3) participate in a panel discussion with the ISAB and the representatives of each of the modeling approaches. The ISAB asked the scientists to address the following questions in their written responses:

1. What is the purpose of your modeling effort? What questions or problems were your models designed to address?

2. Summarize the major conclusions of your modeling effort relative to the four H's: habitat, harvest, hydropower, and hatcheries. Specifically, state your model's conclusions relative to the following:
   a. The efficacy of dam breaching or draw down to natural river levels for delisting ESA species and restoring diverse and productive populations of native fishes throughout the Columbia River Basin?
   b. The efficacy of hatcheries for delisting ESA species and restoring diverse and productive populations of native fishes throughout the Columbia River Basin? Consider hatcheries that mitigate for lost habitat and those used to supplement depleted stocks.
   c. Allocation of harvest and harvest levels needed to delist ESA species and restore diverse and productive populations of native fishes throughout the Columbia River Basin?
   d. The efficacy of restoration of tributary and mainstem habitat for delisting ESA species and restoring diverse and productive populations of native fishes throughout the Columbia River Basin?

3. What kinds of information or data are needed to run your model?

4. What are the strengths and weaknesses of your model?

5. What are the assumptions of your model?

6. How does your model address uncertainty?

7. All models make predictions. Why do you think your model's predictions are accurate?

8. How does your modeling effort relate to or contrast with the other modeling efforts?

9. What advice would you give decision-makers on how they should use your model to support decisions regarding salmon recovery in the Columbia River Basin?

The ISAB held a meeting with the principal scientists on May 3, 2000. We asked them to focus their presentations on the following areas:

1. Provide a brief overview of the analytical approach. Include the kinds of primary data that are needed, quantities derived from the primary data, and conceptually take us through the analytical steps in your approach.

2. Discuss the major assumptions of the approach.

3. Address how the approach deals with uncertainty.

4. Discuss how you believe the analytical approach should be used to support decision-making.

Subsequent to the ISAB's meeting with the scientists, we met individually with the following regional decision-makers on June 7 and 8, 2000: Lori Bodi (Senior Advisor for Fish and Wildlife at the Bonneville Power Administration), William Stelle (then Regional Administrator, National Marine Fisheries Service), Robert Lohn (Director of Fish and Wildlife Division, Northwest Power Planning Council), Donald Sampson (Executive Director, Columbia River Inter-Tribal Fish Commission), and Roy Hemmingway (Salmon Advisor, Oregon Governor's Office). The ISAB asked the decision-makers to address the following questions at the meeting:

1. How do you believe the results produced by the analytical frameworks and other scientific information should be used in the decision-making process?

2. What kind of information do you think the analytical frameworks should provide to best inform decision-making?

3. Do you believe communication between scientists, particularly modelers, and decision-makers need improvement, and if so, how could it be improved?

4. All the analytical frameworks contain a high degree of uncertainty, in part due to the lack of high quality data. Do you believe the level of uncertainty is being adequately communicated to you? How do you incorporate scientific uncertainty in the decision-making process?

5. The five analytical frameworks represent different approaches to somewhat different problems. What is your view of the benefits and detriments for decision-making of having alternative analytical frameworks? From your perspective, do you view the existence of alternative analytical frameworks and results that sometimes conflict as an impediment to decision-making? In making decisions, will you give more weight to some or one of the frameworks over the others? If so, why?

6. What could the ISAB include in its review of the analytical frameworks that would make the review most useful to you?

The ISAB did not request the decision-makers to provide written responses to the questions.

This report is a synthesis of the answers to the questions posed by the ISAB and subsequent discussions with the principal scientists and decision-makers.

Models in Columbia River Basin Salmonid Management

Salmon Science and the Salmon Recovery Effort

The popular press has suggested that Columbia Basin salmon management constitutes a hugely expensive "failure" of "salmon science." We will not comment here on the difficult and inflammatory question of whether the recovery effort, to date, should be judged a failure. We will take this opportunity, however, to correct the misperception about the role of science in the recovery effort. Put most simply, the conventionally cited huge amounts of money spent on the recovery effort were not spent on "salmon science." The bulk of the money was spent, and continues to be spent, on management actions. Historically, the role of science in the process has not been large. Some of the management actions lack a strong scientific basis, and the results of some actions have not been monitored with enough thoroughness to determine their effect.

Overall, two general conclusions are evident. First, the science that was done has contributed substantial knowledge. Second, more science would have contributed more knowledge. There are now prospects for increasing the investment in science; the challenge will be to direct that investment at the right priorities.

The ISAB has reviewed the major modeling approaches that bear on Columbia Basin salmon management. While these approaches constitute "large" modeling projects by the usual standards of ecological modeling, they require a very small fraction of the cost of the recovery effort. The ISAB believes that the investment in modeling and data collection and analysis in the Basin should increase at least in proportion to the desperately needed increase in investment required to establish an acceptable monitoring program.

Primer of Ecological Modeling

Ideally, ecological models, like models of any complicated system, constitute a way of organizing and communicating information. When modeling is done thoughtfully, it provides a systematic and objective way of predicting what we can predict reliably, and identifying what we cannot predict reliably. When the predictive power is slight--as is the case when models are applied to many of the current salmon management questions--a good model serves the very important roles of generating hypotheses and pin pointing the crucial gaps in information. Knowledge about the information gaps provides a valuable guide for setting priorities for new data collection, and it suggests new experiments to resolve the critical uncertainties.

Ecological models provide a means of examining the combined implications of sets of assumptions about mechanisms and sets of measurements on the system of interest. The motivation for this exercise may vary. The motivation is often quite different in academic and management contexts.

In an academic setting, the greatest interest usually centers on use of the model to "test" the assumptions, in order to gauge the current state of theoretical understanding. This is generally carried out by quantifying indices of internal consistency between the data and the relationships implicit in the assumed mechanisms. Often, the goal will be to compare alternative sets of assumptions, to see which are more likely to be true. These academic uses of models are tolerant of very large discrepancies between the model and reality. The model, by design, is often highly idealized, in order to shed light on particular mechanisms of interest, whereas the reality that is the source of measured data is expected to be far more complicated. The mismatch is treated as "noise," that can be largely ignored provided it is not so large as to obscure the contrast between alternative sets of assumptions.

In a management setting, the greatest interest usually centers on the use of models to make predictions -- generally these are predictions about the consequences of alternative interventions that are under consideration. Here, it is taken for granted that the model is an approximation, and probably an oversimplification, and that the actual outcomes will not exactly match the predictions. But the consequences of the decision will result from the actual outcomes. For this reason, discrepancies between the model and reality are of crucial concern. Owing to limitations of our theoretical knowledge and limitations of our available data, these discrepancies may be large, and unavoidable. Because the predictions of the models become a basis for management decisions, the discrepancies between the model and reality cannot just be dismissed as noise. The discrepancies need to be analyzed very carefully to provide a characterization of the uncertainty of the predictions.

The point of the uncertainty analysis for a management model is two-fold. One goal, which should be obvious, is to reduce the uncertainty to the extent that is feasible (even if this is achieved at the expense of loss of elegance or loss of a clear relation to academic theories). The second goal, which is a little less obvious, is to quantify the uncertainty as accurately as possible, in order to provide a rational basis for bet-hedging and setting margins of error in the use of the predictions in a decision process.

Characterization of uncertainty in predictive use of a model is inherently probabilistic, and is carried out by statistical procedures. The quantification of uncertainty is reported in terms of measures of the spread in a probability distribution of the discrepancy between prediction and actuality. The reason for the focus on probabilistic characterization is that deterministic discrepancies can simply be absorbed by calibration. If we know that a prediction is always 10% high, we simply adjust our prediction accordingly. Uncertainty is concerned with the component of the discrepancy that is not so consistent. In other words, uncertainty characterization is concerned with determining the probability that the prediction will be high, or low, by any stated amount. As a technical shorthand notation, the uncertainty characterization is often communicated in terms of the mean or variance of a mathematically defined probability distribution.

The Theory of Ecological Modeling

The "classic" paper on the theory of ecological modeling is R. Levins (1966) " The strategy of model building in population biology." Amer. Sci. 54:421-431. This paper, however, was written from a perspective of academic science, and is not especially concerned with management, so it is only part of the story. It neglects statistics and the quantification of uncertainty. For this reason, this classic paper will provide insight on what is good about CRI, but not about its shortcomings; conversely, it will explain the worst weaknesses of PATH but will neglect PATH's strengths; and it will be totally at a loss for dealing with either the strengths or weaknesses of EDT and ICBEMP-BBN, since expert systems and GIS had not yet been developed at the time the paper was written.

Briefly, a practical update of this theory of ecological modeling would recognize that "scientific model" means many things to many people. So much so, that the intention to classify and analyze all the possibilities is rather daunting. It may be helpful to try to maintain a pragmatic viewpoint and focus on models in a very particular context.

In the present context, we are interested in comparing and evaluating a set of models whose primary purpose is environmental decision making-- more particularly, decision making about big budget interventions and big budget data collection programs, under conditions of high uncertainty and in a very large and complicated ecosystem. Thus, these models are being used to make predictions under circumstances in which we cannot reasonably expect the predictions to be very accurate. To use these models and their predictions intelligently, we have to approach this use in the spirit of intelligent gambling. The first rule of gambling is that to place intelligent bets, you have to know the odds. In the present context, this means that we would need to know, in a quantitative way, how good the predictions of the respective models are. This knowledge is invariably probabilistic.

In other words, "knowing how good the predictions are" means being able to state that if the model predicts X, we have assurance that with probability Y the true outcome will be within plus or minus Z of the prediction X. "Knowing how good the predictions are" means actually having solid numbers for Y and Z, given X. When we know how good the predictions are, in this sense, we can make sensible decisions in light both of the prediction (X), and the probability (Y) that the actual outcome will not be what was predicted, but will deviate by a stated amount (Z). This probabilistic knowledge lets us hedge our bets and spread the risk of being wrong in proportion to the probability of being wrong and the costs of being wrong.

So, the worth of a model in a decision-making context depends on knowing how accurate its predictions are. All other things being equal, we of course prefer a more accurate model to a less accurate model. But if the model is substantially inaccurate, as is likely inevitable in Columbia River salmon management, knowing more about quantifying the inaccuracy is actually more important than making marginal improvements in the accuracy. This is a difficult concept for many people to come to grips with. The lack of understanding on this point often leads to distorted priorities in an overemphasis on "research to reduce uncertainties" and irrelevant controversy about which estimate is "best."

Quantification of the accuracy of the model predictions is generally accomplished by summarizing the fit of model predictions to actual measurements of the predicted quantities (See Appendix I). This is very technical; there are several ways of doing it, and there is not a clear professional consensus on which way is best. Regardless of the mathematical details, there is a known list of requirements for success in the endeavor. Among the most important requirements are the necessity for knowing the precision of the data themselves, a careful identification of actual data versus derived or surrogate quantities, and a rigorous separation between the data used to tune or initialize the model and the withheld data that are then compared to predictions. This last requirement is surprisingly tricky, and depends on a tractable and transparent model structure and a fairly deep understanding of spatial and temporal correlation patterns in the data. A final important and subtle consideration in quantifying the accuracy of model predictions is correcting for the fact that the statistical procedure is inherently retrospective, while our interest usually is in the accuracy of predictions about the future.

Approaches to Modeling in the Columbia River Basin

There are now at least three distinct philosophies of modeling represented in the models available for decision support in Columbia Basin salmon management. Typically, scientists are well trained in at most one of the philosophies. These philosophies are:

1. Decision-analysis, embodied most clearly in PATH-FLUSH and PATH-CRiSP,
2. Statistical, embodied most clearly in CRI, and
3. Expert-system, embodied most clearly in ICBEMP-BBN and EDT.

A more detailed description of the models is given in Appendix II.

Of course, the actual models are not 100% pure manifestations of the respective philosophies, but still each model clearly relies much more heavily on one approach than others. In addition, the strengths and weaknesses of the various approaches do influence the usefulness of the various modeling efforts.

Of the three approaches, the decision-analysis approach is most closely directed at providing management advice, and it is the most formal about factoring uncertainty into the analysis. If this modeling approach is successful, it has the potential to be the most useful to decision-makers.

The decision-analysis approach, however, is very difficult to implement successfully. Its success depends crucially on the engagement of the actual decision-makers in framing the questions that need to be answered, identifying the management options that are under consideration, and in defining the "values" put on the various possible outcomes. The decision analysis approach also requires clear communication between the technical analysts and the decision-makers, including communication about complicated matters of risk, probability, and uncertainty. Such engagement and communication is difficult to achieve in the institutional setting of Columbia Basin salmon management, where there is so much fragmentation of decision-making authority.

The statistical approach is scientifically the most classical of the three, and can operate with a large degree of detachment from policy. It proceeds by testing hypotheses and estimating life history parameters with available data. This has the advantages of clarity, rigor, and empirical objectivity. The limitation of this approach is that the scope of the questions that can be answered with satisfactory conclusiveness is restricted by the availability of data. In a domain that is data-poor, too many pressing questions may go unanswered. This approach may be scientifically correct, but it does not address the needs of the managers who recognize that "no decision" is still a decision.

Expert-system approaches fill gaps in the data with expert opinion. In the context of the salmon problem, expert opinion allows consideration of the most concrete menu of specific options for actual management. Expert opinion is, admittedly, a weaker basis for scientific prediction than is a mathematical relationship validated with an archive of quantitative empirical data. It is important to recognize, however, that at the level of spatial resolution and environmental detail required to make salmon management decisions, there are no available validated mathematical formulae for predicting reliably the effects of management actions on salmon, and there is no adequate data archive for deriving such formulae.

In this light, the expert-system approach may well be a reasonable and practical method for providing tentative answers to some management questions that do need to be addressed quickly. There is a need, now, to make quite specific assessments about a broad spectrum of possible management interventions, to decide which interventions are worth trying, and to decide where, and on what scale, they should be implemented. It is clear, however, that the tentativeness of the answers from the expert-system approach needs to be kept in mind when these answers are factored into management decisions.

When there is a need to make decisions in the face of significant uncertainty, there are valid uses for both secure information and more speculative information such as that derived from expert opinion. In the short term, speculative information is the only option for filling the gaps in the secure information. The two kinds of information should serve different roles in the decision process, and things will go badly if those roles get confused.

Basically, the speculative information generates hypotheses, whereas secure scientific information accumulates through the careful testing of hypotheses and estimation of critical life history parameters. The two can work together if the hypotheses and estimates are used as a basis for designing and implementing management experiments. Monitoring of the results then provides data to test the hypotheses or estimate new parameters. In this way, the expert system and statistical approaches can be complementary, and decision-analysis can optimize the mix.

Prudence demands that we perform management experiments on such a scale that we can afford the consequences of any probable outcome. The contribution of decision-analysis is the balancing of the prospects for each experiment: the value of the desired outcomes, the costs of the undesired outcomes, and the value of the secure information that will be obtained in either case.

For decision-makers to accept scientific advice on the merits of experimental management, the nature and extent of the uncertainties have to be explained in language that is both realistic and vivid. It is not a matter of attempting to teach the decision-makers to like uncertainty -- why should anyone do that? The important thing is learning to cope with unavoidable uncertainty. This is especially important when the uncertainty is large, as is common in environmental problems.

Review of Models: Results versus Approaches

Most of the models we reviewed are, to some extent, works in progress. This has important implications for interpretation of our comments about these efforts. Our comments for each model, accordingly, apply to the state of the model at a particular stage in its development, that is, at the time when we investigated it. To the extent that the developers of the models view their own efforts as ongoing, rather than finished products, we might best frame our review as comments on the respective modeling "approaches" rather than a judgment on a specific set of results.

Roughly ranking the models according to the degree of finality of the effort, it seemed that ICBEMP-BBN was presented as the effort that had come to the most definitive closure, in the sense of defining a question, attempting to answer it, and delivering the answer for the consideration of the decision-maker who had requested it in the first place. ICBEMP-BBN still has the potential for being updated with new data and for further refinement, but at least for the present phase of its use in a decision making process, its developers and users are treating it as being completed.

The PATH effort had the second greatest degree of closure, in that a set of PATH results was delivered for consideration by NMFS in developing their comments for the Anadromous Fish Appendix to the Army Corps of Engineers draft "Lower Snake River Juvenile Salmonid Migration Feasibility Study." In addition, PATH has ceased to exist as an organization within the past year, but some of the component models used in PATH continue to evolve, most notably CRiSP at the University of Washington Columbia Basin Research program, and continuing analytical efforts by the state and tribal fish and wildlife agencies (STUFA) directed primarily at smolt passage and smolt to adult survival, and PATH-EM.

We have witnessed two presentations of CRI, spaced about a year apart, with substantial changes in CRI over that time period being apparent. CRI explicitly represents itself as an "approach" rather than a specific product or model, and both briefings described ongoing analyses and plans for further development. We reviewed in some detail a very early draft of the CRI work intended for inclusion in the Anadromous Fish Appendix.

The EDT effort and the CRITFC model seem to have achieved the least closure. In both cases, the results shown to us were presented as demonstrations of what to expect. In the case of EDT, the planned systematic comparison of the model predictions against empirical data had not yet been carried out. The Council has chartered the Regional Assessment Advisory Committee (RAAC) to manage an effort to compare model predictions against empirical data and to advise the Council on regional analytical needs as part of the Council’s subbasin planning effort. Notwithstanding the tentative nature of the EDT results that were presented to us, the basic underlying model and the nature of the modeling approach were distinctive and clearly presented. The CRITFC model was presented in a form that demonstrated the model structure, but some parameter values used in this model were based on estimates that were known to be outdated, and plans for further work were described.

Comparison of the Analytical Approaches

Summary of the Written Answers to the ISAB's Questions

Purpose of Model.	
ICBEMP-BBN	Compare relative trends in aquatic habitat and population status of six salmonid species among three Federal land management alternatives.
EDT	Compare a set of comprehensive visions (events and actions affecting animal populations and their habitat) in terms of the performance of salmon populations. EDT estimates cumulative effect of one or more individual actions.
CRI	Synthesize data and predict trends, dynamics and risks of threatened and endangered salmonids on the West Coast of North America.
PATH - Flush	Evaluate management options intended to lead to recovery of Snake River listed salmon and steelhead stocks. Only Snake River ESA species are considered.
PATH - CRiSP	Evaluate the effectiveness of a limited number of Snake River salmon recovery actions including breaching the lower four Snake River dams, increased transportation, and reduced harvest.
PATH-EM	PATH Experimental Management (EM) analyzes trade-offs between biological effects of actions, i.e., how actions influence persistence and recovery, for Snake spring/summer chinook. Emphasize how long one would need to monitor population dynamics to detect changes.
CRITFC COHORT	Provide an easy-to-use interface designed to allow users to test salmon recovery options. Provide the essence of existing models for the following factors in a single package: ocean harvest analyses, in-river harvest, PATH downstream passage, and EDT subbasin habitat.

Data Needs of Model.	
ICBEMP-BBN	Predictions or GIS coverages of 6th code HUCs on Federal Lands for: road density, grazing intensity, ground disturbing activities, exotic species, etc. Many of these are extrapolated values, not primary data.
EDT	Habitat information at the HUC-6 level on some 40 habitat parameters. Biological information is required at the population level (rules relating survival to habitat conditions). Many of these “data” are extrapolated values, not original primary data.
CRI	Analyses start with time series of spawner counts to estimate recruits per spawners or annual rates of population growth. Data on the age structure of returning fish are also needed. Many of these 'data' are extrapolated values, not original primary data.
PATH - Flush	Run reconstruction requires: redd counts, dam counts of adults, harvest rates, spawner age composition, hatchery brood removal, and hatchery natural spawner abundance. Other data include: flow rates, travel times, reach survival estimates, passage indices, etc. Many of these 'data' are extrapolated values, not original primary data.
PATH - CRiSP	CRiSP passage model predictions on smolt survival require data on hydrosystem operations, water quality conditions and stock release information.
PATH-EM	Annual estimates of spawning escapement and age estimates, past harvest rates, upstream survival rates, future harvest rates, and estimates of the effects of actions on life-cycle survival (e.g., correlations between spawning escapement and parr to smolt survival estimates). Many of these 'data' are extrapolated values, not original primary data.
CRITFC COHORT	Subbasin habitat data such as temperature and sediment data, mortality rates at various life history stages (such as overwintering, estuary/early ocean, and annual ocean mortality rates), as well as harvest rates. The data needed are those necessary to calculate the instantaneous mortality rate at each life stage. Many of these “data” are extrapolated values, not original primary data.

Strengths of the Model.	
ICBEMP-BBN	Logically explicit and transparent. Results are quantifiable and spatially explicit. The framework is flexible allowing incorporation of both quantitative and qualitative information; analyses can be replicated and assumptions can be tested to determine relevance to the results.
EDT	EDT accounts for cumulative effects of factors such as spatial temporal interactions, all attributes, competition, and predation effects. Density dependent factors are included. It translates combinations of actions at any scale into biological performance responses (population productivity, abundance, and life history diversity).
CRI	CRI is simple, relatively easy to duplicate, and responsive to new and/or updated data. CRI is attempting to deal with observation error.
PATH - Flush	PATH is a cooperative, multi-agency process with peer-review by external scientists. The decision analysis approach made it unnecessary to have complete agreement to determine which actions had the best chance of achieving recovery over the range of uncertainty.
PATH - CRiSP	The CRiSP passage model is calibrated with all the available current data.
PATH-EM	PATH-EM is a Ricker population projection model of Recruits vs. Spawners. Management regimes can be as simple or complex as desired by the model user, effecting all stocks permanently (e.g., drawdown) or different stocks in different ways over time (e.g., different nutrient enrichment schedules for different stocks and years).
CRITFC COHORT	The model has a simple user interface allowing anyone to run the model. It is one of the few models to include the potential effects of habitat change on population growth

Weaknesses of the Model.	
ICBEMP-BBN	Limited data and knowledge at relevant scales forcing substantial reliance on expert opinion, and surrogates for key elements or processes. Results can be viewed only as relative trends among alternatives.
EDT	Lack of ground truthing of input data and peer review to ensure that rules are consistent with current information and knowledge. Need modeling of genetic effects to be expanded. When applied to the entire Columbia Basin EDT takes hours to run and requires lots of computer space.
CRI	Pessimistic, if one assumes that ocean conditions will soon improve. CRI is not a formal decision theoretic approach.
PATH - Flush	The complexity of analyses in PATH was not transparent to many individuals. There is limited data for building models, testing hypotheses about the past, and testing hypotheses about the future.
PATH - CRiSP	There is no way to characterize differences in survival among fish of a release group for different passage routes or because of pre-smolt conditions. The PIT tag studies in the tributaries above the hydrosystem do not support a simple travel time survival relationship as is contained in the CRiSP and FLUSH models.
PATH-EM	There is strong reliance on spawner-recruit estimates, which may contain systematic errors. Estimates of an action’s effects are needed as model inputs.
CRITFC COHORT	There are no density dependent effects, no stochasticity, no time lags built in for hydrosystem or habitat changes, does not allow changes in flow, and population sizes are not projected into the future.

Assumptions required by the Model.	
ICBEMP-BBN	The assumed conditional probabilities adequately represent key relationships, processes, and interpretations of management direction. Proposed restoration and management will proceed as described. Local habitat, biological potential, and ocean and migratory conditions are the primary determinants of species status.
EDT	EDT is a flexible, expert (cause and effect) system that begins with a detailed habitat description and links it via assumptions to ecological function and biological performance of specific species. Two parameter density-dependent survival functions are assumed for each life cycle segment (typically the Beverton-Holt equation is used).
CRI	No density-dependence in current dynamics. No directional trends in the variation in recruits per spawner.
PATH - Flush	The prospective analysis correctly quantifies the range of possible future conditions. Under each management option, the hydro-regulation model correctly computes mean monthly flows, the passage models correctly compute passage survivals and the Bayesian life cycle simulation model (BSM) correctly generates a range of spawner abundances.
PATH - CRiSP	CRiSP assumes survival is a function of travel time and environmental water quality including temperature and total dissolved gas levels.
PATH-EM	Future conditions will be like 1978-1994 absent management intervention and the estimates of an action's effects (which are model inputs) are accurate.
CRITFC COHORT	Spawning and rearing are evenly distributed throughout currently used habitat regardless of habitat quality. Habitat improvements are applied first to the lowest quality habitat. A per-dam mortality rate of 17% is assumed. John Day drawdown reduces mortality by ½ of a dam. Hatchery fish upon release are treated as being identical to natural origin fish with the exception of a 50% post release mortality applied to the hatchery fish.

Conclusions with respect to Dam Breaching.	
ICBEMP-BBN	Not useful for evaluating the efficacy of dam breaching or drawdown. Models reflect the expert's beliefs that dams have a dominant influence on status of anadromous salmonids.
EDT	Breaching dams was effective in restoring mainstem habitats to the benefit of chinook performance. Dam removal provided significant benefits to listed Snake River stocks, but less or minor benefit to Upper Columbia River stocks.
CRI	Dam breaching would need to increase ocean/estuarine survival by 100% or more in order to ensure positive growth rates large enough to mitigate extinction risk.
PATH - Flush	In most cases, the “natural-river” options met the survival standards under the most pessimistic assumptions. None of the “transportation” options met the recovery standard, except under very optimistic assumptions. The absolute values of PATH population projections are likely optimistic.
PATH - CRiSP	Drawdown was better than transportation as a recovery measure for spring chinook while harvest was a possible recovery measure for fall chinook. New data now challenge the PATH conclusions.
PATH-EM	Effects of dam breaching, e.g., effects of drawdown of four Lower Snake projects, have been modeled by PATH-EM. Some preliminary results are in the PATH-EM Report, but were not given during the presentation.
CRITFC COHORT	CRITFC has not reached final conclusions on the questions posed. Based only on an aggregate of the Middle Fork Salmon R. spring chinook they conclude: 1) spring chinook populations cannot be rebuilt without dam breaching, and 2) survival gains from other factors are needed to allow the population to rebuild, even with dam breaching.

Conclusions with respect to Hatcheries.	
ICBEMP-BBN	Efficacy of hatcheries was not addressed.
EDT	Conclusions were not given. EDT brackets alternatives by using a range of assumptions about effectiveness of hatcheries. The initial assumption was that hatcheries fulfilled their promise and were comparably effective as natural populations.
CRI	Because basic data needed to assess the impacts of hatchery fish are lacking, including reproductive output of wild spawning hatchery fish, it is impossible to obtain answers to many questions regarding the impact of hatchery fish on wild fish.
PATH - Flush	Preliminary results, based on limited data, suggest that artificial propagation of spring/summer chinook has not significantly contributed to declines in wild populations of Snake and upper Columbia spring/summer chinook.
PATH - CRiSP	PATH did not address hatcheries in a scientific manner.
PATH-EM	Effects of hatcheries on wild populations, e.g., effects of reducing Snake hatchery releases, have been modeled by PATH-EM. Some preliminary results are in the PATH-EM Report, but were not given during the presentation.
CRITFC COHORT	CRITFC has not reached final conclusions on the questions posed. Considering only demographic effects, hatcheries are an effective tool for maintaining, reintroducing, and rebuilding existing populations.

Conclusions with respect to Changes in Harvest.	
ICBEMP-BBN	Allocation of harvest levels was not addressed.
EDT	Harvest reductions had little impact on spring chinook stocks but were important to the abundance of fall chinook. EDT has not attempted to assess impacts of harvest rates on ESA status.
CRI	Reducing harvest would allow three ESUs (Fall Chinook in the Snake River, Upper Willamette Chinook, Lower Columbia Chinook) to grow. Harvest reductions by themselves could not esure growing populations in the remaining ESUs.
PATH - Flush	Declines in survival of Snake River Basin populations occurred at the same time as substantial declines in harvest rates, contrary to the assumption that harvests are significant contributors to declines in survival of Snake River Basin populations after 1974.
PATH - CRiSP	The mixture of hypotheses were optimistic in their estimates of future productivity of stocks, so harvest reductions predict optimistic recovery probabilities.
PATH-EM	Effects of different harvest strategies have not been modeled by PATH-EM. Future harvest strategies are required as input for running the model.
CRITFC COHORT	CRITFC has not reached final conclusions on the questions posed. Limited analysis indicates that total elimination of all harvest, by itself, is not sufficient to maintain or rebuild spring chinook populations in the Snake River or above Priest Rapids dam.

Conclusions with respect to Tributary and Mainstem Habitat Improvement.	
ICBEMP-BBN	All alternatives were projected to improve habitat on FS/BLM lands and to maintain current strong populations with some rebuilding of depressed populations. The preferred alternative showed the largest relative increases.
EDT	Breaching dams was effective in restoring mainstem habitats to the benefit of chinook performance. Land-use changes and practices in tributaries were, over the long term, effective in restoring abundance and distribution of tributary populations.
CRI	In regression modeling, salmon productivity was found to be more related to: 1) good quality riparian zones in non-forested areas, 2) low water temperatures, and 3) low gradients, as contrasted to the number of dams passed.
PATH - Flush	The significant decline in Snake River Basin populations after 1974 does not coincide with habitat degradation. Degradation in habitat occurred mostly prior to 1974 and spawning and rearing habitat for some of the Snake River Basin populations has remained in good or pristine condition.
PATH - CRiSP	PATH had no essential contribution to the effect of restoring diversity of habitat in tributaries or the mainstem. Assessments for restoring or creating mainstem fall chinook habitat were approached in a cursory manner.
PATH-EM	Effects of specific tributary and mainstem habitat improvements can be modeled by PATH-EM. For example, effects of improvement in rearing habitat (increasing egg-smolt or parr-smolt survival via carcass or nutrient enrichment) have been studied. Some preliminary results are in the PATH-EM Report, but were not given during the presentation.
CRITFC COHORT	CRITFC has not reached final conclusions on the questions posed.

How is uncertainty addressed?	
ICBEMP-BBN	Key relationships in the networks are represented by conditional probabilities. The model outputs were expressed as the probability of future habitat capacity and salmonid population status.
EDT	The implications of uncertainty are addressed through sensitivity analysis (e.g. by presenting "best case-worst case" outcomes of actions). EDT is a steady-state model that does not incorporate stochastic elements.
CRI	Compute confidence intervals for estimates of annual population growth. In case of little data, CRI uses an upper and lower bound for parameters or perform numerical calculations over a wide range of potential values.
PATH - Flush	The Bayesian life cycle simulation model (BSM) was used to carry uncertainties in climate and stock-production relationships through the analysis. The decision analysis approach considered all possible combinations of all relevant uncertainties.
PATH - CRiSP	Uncertainty in PATH models was expressed through a weighting of the hypotheses by outside experts. The degree of certainty in conclusions may have been overstated.
PATH-EM	Summarizing the sampling distribution via confidence intervals assesses uncertainty in parameter estimates. The population projection model respects parameter uncertainty by drawing values from the joint sampling distribution of the estimates.
CRITFC COHORT	The user can conduct a sensitivity analysis on variables that are of concern.

Accuracy of predictions?	
ICBEMP-BBN	The models are accurate in the sense that they faithfully reflect the beliefs of experts as to the influence of land management on aquatic habitats and salmonids.
EDT	EDT's depiction, like that of any model, is imperfect. EDT compares relative effects of different habitat changes on salmon performance.
CRI	Some predictions are accurate, e.g., if downstream survival was increased to 100%, then some populations are still at severe risk. Some predictions are not. CRI depends on relative rank of predictions, not accuracy.
PATH - Flush	Emphasize that relative ranking of actions may be more important than absolute standards. Rank order of actions was insensitive to choice of life-cycle model configurations and passage models.
PATH - CRiSP	CRiSP model predictions for spring chinook hydrosystem survival have been proven accurate by comparisons with PIT tag data.
PATH-EM	The fit of the model has been tested by addition of new index stocks and by excluding/including recent years of spawner/recruit data. Of course, the future is unknown and for example, the model's assumption about low-survival under recent ocean conditions may not hold.
CRITFC COHORT	Population growth rates from the model for spring chinook are of similar magnitude to observed rates estimated independently by others.

Relation to other models?	
ICBEMP-BBN	Each modeling effort had very different objectives. ICBEMP-BBN provided a broad scale but more intensive analysis of current and proposed management of FS/BLM lands than the other models. ICBEMP-BBN analyses concerning migration survival did not change relative differences of projected effects among the three alternatives.
EDT	The relationship between CRI, PATH, and EDT is complementary. EDT can incorporate the analytical results of the statistics-based PATH and CRI. Both EDT and ICBEMP-BBN rely on expert opinion, but EDT gives finer scale analyses. EDT might complement the CRITC model by expanding its analytical capabilities.
CRI	CRI is data based, focusing on annual rate of population change and the risk of extinction for eleven different ESUs. PATH studied two ESUs and focuses on hydropower. EDT is an expert system summarizing effects of habitat descriptors, but does not incorporate much fish population dynamics. ICBEMP-BBN seems to be primarily a habitat-focused analysis.
PATH - Flush	Concentration on Snake River stocks allowed PATH to perform a comprehensive analysis of two data rich stocks (spring/summer chinook and fall chinook). CRI, with its emphasis on rate of growth, is charged with analyzing all ESA listed anadromous stocks, regardless of the amount or quality of data available. EDT models both anadromous and resident fishes at basin-wide or province-wide scales.
PATH - CRiSP	PATH synthesized much of the past data. The other modeling projects have used the PATH data analyses as a foundation for their work.
PATH-EM	Compared to CRI, PATH-EM allows a more systematic exploration of trade-offs among population risk measures, experimental design, and knowledge gained. For example, PATH-EM allows the growth rate of populations to decrease with increased population size.
CRITFC COHORT	We have attempted to represent the general behavior of PATH, PSC, and TAC analyses without much of the complexity and detail.

How should the model be used to support decisions?	
ICBEMP-BBN	The ICBEMP-BBN model was specifically created to help decision-makers select a management plan for FS/BLM lands. It should be used in concert with other models and information to weigh the merits of possible decisions affecting management of other public and private habitat, hydropower, hatchery operations, and harvest.
EDT	Models highlight risks and tradeoffs of alternative actions. No model does it all for most decisions. Use EDT in conjunction with other models such as the CRI, PATH, and CRITFC tools. Take advantage of the strengths of each model system and appreciate the different perspectives afforded by each. Do not use models as analytical bludgeons in political warfare.
CRI	Decision-makers might be well-served by drawing on all of the available analytical tools. EDT and CRI are not 'dueling' models; in fact, they attempt to predict totally different response variables. PATH and CRI could be profitably combined, or at a minimum, learn from one another. ICBEMP-BBN seems to be primarily a habitat-focused analysis.
PATH - Flush	The decision analysis of PATH showed which actions were most likely to benefit the stocks under the widest range of assumptions. Analyses using the same data and the Leslie matrix model advocated by CRI give strikingly similar results about the relative effectiveness of the proposed alternatives in preventing extinction and achieving recovery of listed Snake River stocks.
PATH - CRiSP	Neither CRiSP, nor any other passage or life cycle model now available, is adequate to address the issue of delayed mortality or survival of fish above the hydrosystem. The models need to be further developed and new experiments need to be planned to characterize the impacts of passage experience on fish survival.
PATH-EM	Take the individual population dynamics predictions with several grains of salt: changes in climate and other uncontrollable factors make predictions of extinction times, probabilities, etc. very uncertain. However, have confidence in the overall conclusions from the EM modeling.
CRITFC COHORT	This model can be an effective tool for teaching local watershed councils about the impact of conditions outside individual subbasins upon local restoration efforts. The model can be used for broad scale trade-off analyses across the 4-H's.

Synthesis of the Answers to the ISAB's Questions

What is the purpose of your modeling effort? What questions or problems were your models designed to address?

Purposes of the decision support systems (models) are different. Two (PATH-FLUSH and PATH-CRiSP) are primarily designed only to evaluate Snake River salmon recovery actions including breaching the lower four Snake River dams, increased transportation, and reduced harvest. Two (ICBEMP-BBN and EDT) are primarily designed to evaluate large-scale habitat management alternatives across the landscape of the Columbia River Basin and three (CRITFC, CRI and PATH-EM) are primarily for prediction of trends in populations of salmonids based on counts of fish and rates of change in population parameters.

What kinds of information or data are needed to run your model?

Data needs are different, because the purposes of the models are different. PATH-FLUSH and PATH-CRiSP models concentrate on estimating survival based on flow and other data as fish enter the lower Snake River. ICBEMP-BBN and EDT concentrate on predicted effects of habitat changes over large areas based on spatial habitat data. CRITFC, CRI, and PATH-EM concentrate on predicted trends in sizes of populations based on fish counts and rates of change of population parameters (e.g., survival rates) in the presence of various management actions. One characteristic common to all of the models is that information required to run the models includes not only primary data measured by documented procedures, but also derived values each depending on yet other (perhaps undocumented) assumptions and 'models.'

What are the assumptions of your model?

Not only are the purposes and data needs different among the models; the models depend on different assumptions made by the respective developers concerning the state of nature. Some of the key assumptions made are highlighted in Table 1, and the presenters described others (Appendix III). Two of the models, ICBEMP-BBN and EDT, are "expert systems" depending heavily on opinions (i.e., assumptions) expressed by panels of experts. All of the models implicitly assume that future climate and ocean conditions will be like those in the recent past.

What are the strengths and weaknesses of your model?

Given that purposes, data needs, and assumptions are different, it is not surprising that the models have different strengths and weaknesses (Table 1). Some involve complex interactions of physical and biological factors (ICBEMP-BBN, EDT, PATH-

FLUSH, and PATH-CRiSP), others (CRI and CRITFC) depend on relatively simple mathematical procedures that can be easily programmed and duplicated, while PATH-EM fits somewhere in-between. Simplicity and complexity are each simultaneously a strength and a weakness. The simpler models allow relatively more direct point estimates of life history parameters under a given set of assumptions, but lack the ability of the complex models to help understand mechanisms of cause and effect relationships. The more complex models deal better with the comparison of management alternatives, but usually can only rank alternatives on their expected benefits. It would be nice of course to have the advantages of all the decision support systems in one “model”, but this seems to be impossible.

How does your model address uncertainty?

Only one model (CRI) reported an attempt to deal with variation in the data due to sampling (observational error) so that confidence intervals can be placed on predicted values. However, CRI was not totally successful in this attempt and along with EDT, PATH-EM, and CRITFC, performed sensitivity analysis (e.g. presented "best case-worst case" outcomes of actions). The other models, ICBEMP-BBN, PATH-FLUSH, and PATH-CRiSP, utilized probabilistic arguments either by standard procedures or Bayesian methods to assign probabilities to expected outcomes. None of the models address the issue of the feasibility of implementing management actions in the face of social and economic constraints.

All models make predictions. Why do you think your model’s predictions are accurate?

In comparison of major management alternatives, all of the models depend primarily on the relative rank of predictions, not accuracy of the actual predicted numerical values. The rank orders of effects of alternative actions were judged to be accurate by the model presenters.

How does your modeling effort relate to or contrast with the other modeling efforts?

The modeling efforts have very different objectives. Both EDT and ICBEMP-BBN rely on expert opinion to rank effects of habitat changes over large areas, but EDT gives finer scale analyses. CRI, PATH-EM, and CRITFC attempt to predict the trends of stocks of fish, but with different approaches and information. PATH-FLUSH and PATH-CRiSP concentrate on analysis of only two Snake River ESUs. Nevertheless, the general conclusions when faced with the same objectives and when based on more-or-less the same information are often in close agreement with respect to the predicted rank order of effects of management alternatives. As PATH-FLUSH concluded “The decision analysis of PATH showed which actions were most likely to benefit the stocks under the widest range of assumptions. Analyses using the same data and the Leslie matrix model advocated by CRI give strikingly similar results about the relative effectiveness of the proposed alternatives in preventing extinction and achieving recovery of listed Snake River stocks.”

What is the efficacy of dam breaching or draw down to natural river levels for delisting ESA species and restoring diverse and productive populations of native fishes throughout the Columbia River Basin?

There is general agreement among the models that breaching dams is effective in restoring mainstem habitats to the benefit of listed fall chinook and breaching would provide significant benefits to both Snake River fall and spring listed stocks. All models also agree with the obvious and important conclusion that breaching dams on the Lower Snake River will do little or nothing to help the other ten listed ESA species. With respect to the Snake River spring chinook population, there is general agreement among the PATH models that dam breaching or draw down are better than transportation (with the dams in place) as a recovery measure. CRI and CRITFC conclude that survival gains from other factors such as tributary and estuarine habitat improvement are needed to allow the Snake River spring chinook population to rebuild, even with dam breaching. ICBEMP-BBN is not useful in judging the efficacy of dam breaching.

What is the efficacy of hatcheries for delisting ESA species and restoring diverse and productive populations of native fishes throughout the Columbia River Basin?

No satisfactory conclusions were given concerning the efficacy of hatcheries for delisting ESA species. Only EDT and PATH-EM seems to have taken an initial look at modeling the effect of reduced hatchery production (for wild steelhead in the Snake River).

What is the allocation of harvest and harvest levels needed to delist ESA species and restore diverse and productive populations of native fishes throughout the Columbia River Basin?

The effect of changing harvest received varied and limited attention by the modeling efforts. CRI concluded that reducing harvest would allow only three ESUs (Fall Chinook in the Snake River, Upper Willamette Chinook, Lower Columbia Chinook) to grow, a conclusion not much different from the PATH determination that harvests were not significant contributors to declines in survival of Snake River Basin populations after 1974, or the CRITFC note that elimination of all harvest, by itself, is not sufficient to maintain or rebuild spring chinook populations in the Snake River or above Priest Rapids dam. ICBEMP-BBN, EDT, and PATH-EM have not attempted to assess impacts of harvest alternatives on ESA stocks.

What is the efficacy of restoration of tributary and mainstem habitat for delisting ESA species and restoring diverse and productive populations of native fishes throughout the Columbia River Basin?

The two broad scale habitat models, ICBEMP-BBN and EDT, both agree that over the long term improving tributary habitat is effective in maintaining or restoring abundance and distribution of tributary populations. The rather obvious conclusion is supported that breaching dams was effective in restoring mainstem habitats to the benefit of listed Snake River fall chinook. CRI related salmon productivity to characteristics of tributary habitat. PATH, in contrast, concluded that tributary habitat degradation was not responsible for the decline in Snake River stocks following the construction of the lower Snake River dams.

What advice would you give decision-makers on how they should use your model to support decisions regarding salmon recovery in the Columbia River Basin?

The decision support systems have different strengths and weaknesses and policy-makers would be well served by drawing on all of the available analytical tools. Six of the seven presenters implied that their model should be used in conjunction with other models, because in the words of one "No model does it all for most decisions. Use EDT in conjunction with other models such as the CRI, PATH, and CRITFC tools. Take advantage of the strengths of each model system and appreciate the different perspectives afforded by each." The ISAB wholeheartedly endorses this recommendation. Results from a model, given knowledge of its assumptions and quality of data used, cannot decrease the information available to a decision-maker.

Evaluation of the Modeling Efforts

None of the models presently in use in the Columbia River Basin is complete enough to serve as the sole decision support tool for the region. The models available to support decisions in the Columbia Basin serve different functions and all have strengths and weaknesses. The models differ in the problems they were attempting to address, the analytical approaches to the problems, the assumptions underlying each of the approaches, the quantity and quality of the available data, and the rigor with which they deal with the complex life cycles and habitats of the species.

PATH scores high for its attempt to operate in a decision context that was explicit about uncertainties, and for attempting to formally quantify uncertainty in a reasonably sophisticated way. PATH's weakness was in the very narrow range of factors that it took into consideration, and in the tortuous model structure that posed an obstacle to diagnosis of model behavior. CRI, by contrast, scores very high for clarity and tractability of model structure, but it lacks statistical treatment to quantify uncertainty. CRI encompasses a broader range of factors than the original PATH process, but it still appears deficient in necessary detail for habitat variables and spatial and temporal structure. Both CRI and PATH rely on derived quantities in place of actual data, in ways that confound a proper uncertainty analysis. CRI clearly is investing heavily in attempts to remedy these deficiencies.

EDT scores very high marks for comprehensiveness of factors taken into account, and for spatial and temporal detail in the data it uses. It is not yet clear whether EDT is sufficiently rigorous about maintaining the separation between actual measurements and derived quantities, or in quantifying the uncertainty of the derived quantities. The predictive modeling component of EDT is essentially an expert system -- this is a systematic amalgamation of the judgment of a panel of experts that then gets coded into a set of quantitative relationships that can operate on data input to generate predictions. In principle, the construction of an operating rule set from the expert judgment can be done in a way that is transparent and open to diagnosis. We do not yet know whether EDT has achieved this. The process of eliciting expert judgment may include obtaining opinion about certainty, but that too is expert judgment, rather than a statistical comparison to actual measurements. So it looks as if EDT does not yet include a satisfactory treatment of uncertainty.

ICBEMP-BBN like EDT gets high marks for comprehensiveness and for spatial and temporal detail. ICBEMP-BBN went one important step beyond EDT by formally eliciting expert judgment about uncertainty and propagating this uncertainty through the model predictions. ICBEMP-BBN was constructed as a Bayesian Belief Network, which confers the advantage of explicit representation of uncertainty as probabilities. This in turn offers the potential of a smooth transition from expert system based analysis to statistically based analysis, as more data become available to develop more empirical models of the key relationships between management, habitat, and salmon population responses. Essentially, the Bayesian Belief Network can serve as a prior distribution for classical Bayesian statistical inference, merging the expert opinion with empirical measurements. The Bayesian statistical framework lends itself to value-of-information calculations that can help guide decisions about future monitoring, ground truthing, or experimental interventions to help resolve critical uncertainties. ICBEMP-BBN did not avail itself of this potential, but it is an attractive option that should be considered as EDT evolves toward calibration and validation.

Overall, the region needs modeling efforts that collectively combine the best features of all the models we have reviewed. Features of ICBEMP-BBN and EDT could be used to formulate working hypotheses concerning, for example, causes of salmonid declines and potential effectiveness of management interventions. Features of ICBEMP-BBN could be used to communicate the uncertainty of the working hypotheses. Features of CRI could be used to test hypotheses wherever data were available. Features of PATH could be used to quantify the uncertainty of the tests of the hypotheses and to place recommendations for management in a risk assessment context. Furthermore, CRI might serve as an inspiration for clarity, rigor, and openness.

The models the ISAB reviewed are best at ranking the expected effects of management alternatives. The general conclusions of the models are often in close agreement with respect to the predicted rank of management alternatives when addressing similar problems and using the same data sets. They are not good, however, at giving absolute numerical predictions and they do a poor job of accurately estimating what the policy-makers may need most, namely, a credible scientific analysis of the probability (feasibility) that some measurable degree of salmon recovery will be achieved with any particular management action.

There are two important ways in which scientists can help environmental decision processes to cope better with uncertainty. The first is the purely technical contribution of explicitly quantifying the relevant uncertainty. Statements of the respective probabilities of alternative scenarios are a natural way to communicate uncertainty when the decision is essentially placing a bet about which scenario actually will materialize. Recent progress in practical methods for quantifying uncertainty is an exciting development that deserves to be used in Columbia Basin salmon modeling.

The second important area where scientists can contribute is in helping the decision-makers craft decision rules that get formalized before the analysis is undertaken. Decision rules define the measurements that will be made, the statistical operations that will be performed on the data, and the threshold magnitudes of estimated quantities at specified levels of certainty that will serve as criteria for the decision. Such specifications help remove ambiguity from the way science is used in a decision -- even when there is uncertainty in the data or models. Committing to these specifications in advance helps dispel suspicions that the analysis may be manipulated to achieve a particular outcome.

All the models are severely constrained by lack of data. Modeling controversies in the salmon arena have largely been an unproductive distraction from the real scientific problem of inadequacy of the available data for addressing many of the important management questions. Some of the debate that now centers on competing models could be resolved with the right data. The present paucity of data creates more scope for alternative assumptions in the models. Sophisticated, responsible modeling takes all the plausible alternative assumptions into account with weighting according to their respective concordance with the data that are available. This need not lead to "modeling wars."

The data problem, unfortunately, will not be solved as easily as the modeling problem or the data base management problem. Where scientific leadership and institutional innovation needs to be exercised is in the prioritization, design, and implementation of large-scale monitoring linked to management experiments. There is at present no institutional center of authority for addressing the prioritization, design and coordination issues for large-scale monitoring. Prioritization and design issues inherently presume centralization, so the entrepreneurial "distributed" paradigm that the ISRP has recommended for data management, and that the ISAB is here recommending for modeling, will not solve the data problem.

Decision-makers would be well served by drawing on all the available analytical tools. Decision-makers would benefit by focusing on areas of consensus among the models or the weight of evidence provided collectively by the models. Areas of disagreement among the models may pinpoint uncertainties that require further investigation. In considering how results of models make their way into the decision-making process, it is helpful to recall the roles of models. They provide ways of organizing and communicating information, generating hypotheses, and pin pointing the crucial gaps in information. The modeling efforts are not ends in themselves; they are not final, definitive answers, but rather they are ongoing processes for continuously increasing knowledge.

Moving Forward with Support for Multiple Modeling Efforts

At this point in the evolution of northwest salmon science, there is no compelling reason to fund large collaborative modeling efforts. For the amount of real data that are available, the kinds of models that are actually justifiable really are not that complicated. Small groups of researchers should be adequate to pursue development of the models. And the coherence of vision of a small group may encourage clarity in the resulting model. That coherence is very difficult to achieve as the group gets large.

The past hope in encouraging the development of a region-wide collaborative modeling effort was to achieve "scientific consensus." In fact, the lesson of reflecting on what we have learned in the last period of time is that scientific consensus does not really emerge from regional modeling committees. Scientific consensus is a product of a much larger scientific audience, whose primary route of exposure to the information in question through reading articles, mostly in peer-reviewed books and journals; and where major differences are settled by the results of attempts to duplicate analyses of the same data or attempts to duplicate experiments in the same system.

So, the correct way to proceed with modeling needs is to fund specific groups to undertake specific modeling projects. For any modeling topic that is expected to be in any way controversial, one of the conditions of the contract should be delivery of a version of the report in a form suitable for peer-reviewed publication. The length limitations of this publication format may preclude some important detail, so it is to be expected that there will be "long-form" reports as well (which may include data archives) that should be in electronic form to be made accessible on the Web.

The expectation will be that the reports delivered in a form suitable for peer-reviewed publication actually will be submitted for publication, but it is recognized that the review (and possible revision) and publication process usually involves a lag time of six months to two years. The long lag time should not be used as an excuse not to pursue publication. It is true that managers and administrators often believe that they need the results "yesterday," but it is also true that the same questions come up again and again, year after year, often without satisfactory resolution.

There would be merit to funding several groups to pursue modeling questions. There would even be merit to a degree of overlap, so that the same data sets might be analyzed from somewhat different technical perspectives, which would encourage the evolution of scientific insights. The actual dollar cost of modeling, as a fraction of the overall salmon recovery budget, has been very small, so this is not the place to look for cost cutting.

Use of Models in the Decision Process in the Columbia River Basin

The Science-Policy Interface

For scientific results to enable management interventions that are effective in accomplishing societal goals, there must be not only a belief by decision-makers in the credibility of the science and its relevance to the problems being addressed, but also effective communication between scientists and decision-makers, so that decision-makers are well informed of the consequences of management alternatives. Each of the present collection of prominent salmon-relevant modeling approaches that we reviewed has something useful to offer policy-makers provided the policy-makers understand the logic, supporting evidence, and limitations and assumptions associated with each model. Without that understanding, of course, there is a temptation to use the modeling results in superficial and spuriously definitive ways that can add confusion to the deliberations. In addition, misuse and misunderstanding have the potential to create an atmosphere that undervalues science.

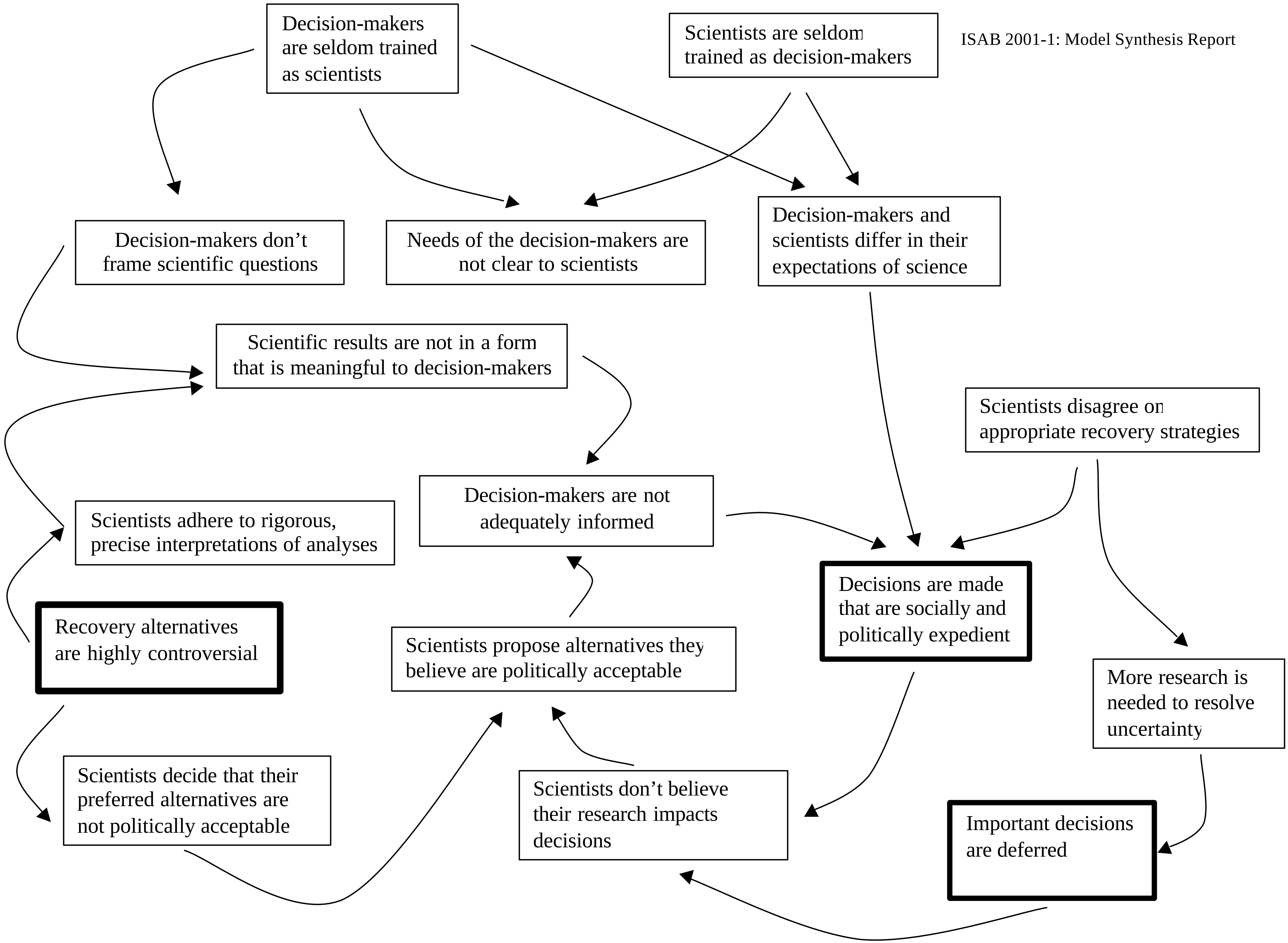
Unfortunately, there is no standing formal institutional mechanism for synthesizing the results or for clarifying the interpretation of the various salmon models for the various policy makers on an ongoing basis. In this connection, that decision-making authority for various aspects of salmon management is as fragmented as the modeling efforts.

The following sketch and accompanying text is our attempt to provide an analysis of potential impediments at the policy-science interface that were revealed in our discussions with decision-makers and scientists. The sketch portrays a system of self-regulating behaviors exhibited by both policy-makers and scientists (Bella 1995, 1996a,b, 1997). The idea is that policy-makers and scientists become "locked up," trapped as it were, in a system of behaviors that form self-reinforcing loops, thus tending to perpetuate the status quo. We do not intend to imply that the patterns of behavior depicted in the sketch are operative under all circumstances.

The behaviors are represented in boxes that are connected by arrows. The incoming arrows describe reasons for the behavior and the outgoing arrows describe consequences of the behavior. Proceed forward or backward along any arrow. If you move forward, say "therefore" and if you move backward say "because." Start at the top left of the sketch.

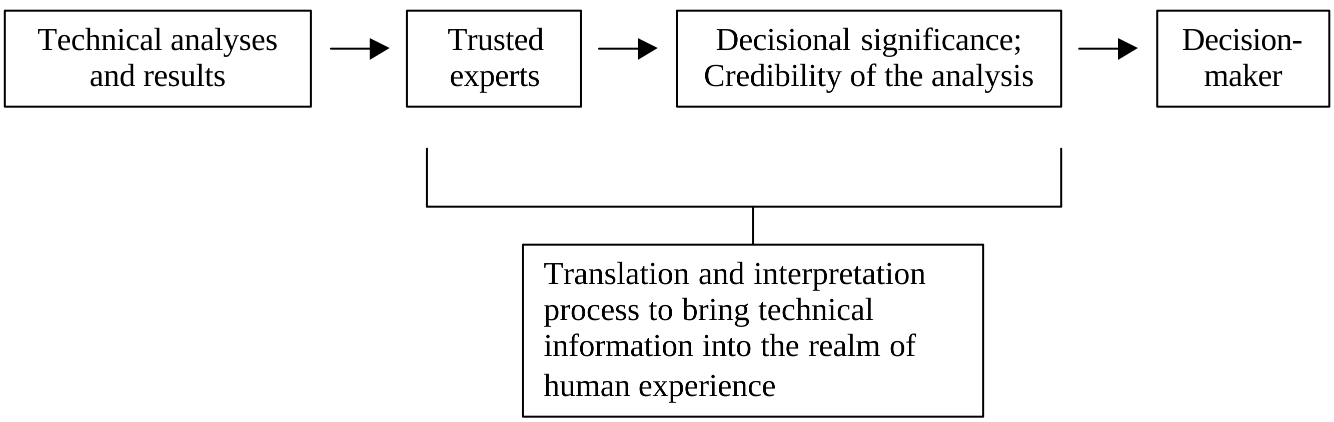
Decision-makers and scientists often not only "speak different languages," but also operate with different interests, goals, and belief systems. Seldom are decision-makers educated as scientists and thus, they sometimes ask scientists to answer questions that are not amenable to a scientific approach or phrase questions in a way that is unclear to scientists. Scientists, in turn, are often not trained and have little experience in the policy-making process in which the legal and societal dimensions of any proposed solution must be considered. Some scientists are simply not interested in the policy implications of their work, often viewing themselves as independent of the policy process, and focusing mainly on conducting scientific research within their area of expertise with the belief that their research results will somehow feed into the decision-making process. Scientists sometimes tend to view the politics of the policy-making process as value-laden, subjective, and sometimes corrupt.

As a result of differences in their background experience, education, and interests, the needs of decision-makers often are not clear to scientists, and decision-makers do not comprehend what science is able to accomplish relative to a particular problem. Thus, the expectations of scientists and decision-makers differ, with decision-makers often expecting science to accomplish, over a relatively short time period, more than it is capable of accomplishing given limitations of data quality and quantity, environmental variability and uncertainty, limited background and experience of scientists, and so forth.


Decision-makers are seldom trained as scientists
Scientists are seldom trained as decision-makers
Decision-makers don't frame scientific questions
Needs of the decision-makers are not clear to scientists
Decision-makers and scientists differ in their expectations of science
Scientific results are not in a form that is meaningful to decision-makers
Scientists disagree on appropriate recovery strategies
Decision-makers are not adequately informed
Scientists adhere to rigorous, precise interpretations of analyses
Decisions are made that are socially and politically expedient
Recovery alternatives are highly controversial
Scientists propose alternatives they believe are politically acceptable
More research is needed to resolve uncertainty
Scientists decide that their preferred alternatives are not politically acceptable
Scientists don't believe their research impacts decisions
Important decisions are deferred

As a consequence of this shortage of meaningful communication, decision-makers are often not as well informed as they could be of the full range of management alternatives and uncertainties. When decision-makers do not frame appropriate scientific questions or fail to make their needs clear to scientists, two things can occur. First, research is performed that does not address critical management questions. Second, technical results are communicated to decision-makers in a form that is not meaningful to them. As an example, technical measures such as the risk of extinction, expressed as the probability of extinction over 24, 48 or 100 years, are perfectly acceptable scientific measures of the chances of extinction, but these concepts and the time frames to which they apply may be difficult for decision-makers to interpret and implement into a defensible decision.

Decision-makers most often rely on trusted experts, who are usually members of their own staffs or independent scientific groups, to assess and interpret, “translate” as it were, results of technical analyses into a form that can feed into the decision process more readily, without seizing the decision from the decision-maker.



Another pathway that leads to poorly informed decision-making could originate when management interventions are highly controversial and have major political, social, and economic ramifications. Scientists sometimes respond to this situation by retreating “into the science”, becoming overly precise and rigorous in communicating the results of their analyses to decision-makers. Scientists act this way because they believe that in doing so the decision-maker will have accurate facts and the scientists themselves will not be held responsible for any errors in interpretation. In other cases, scientists may have so much time and energy invested in a particular modeling approach that they become advocates for the results of the model, losing sight of the uncertainties. Less than well-informed decision-makers should be concerned about their ability to communicate clearly and justify their decisions to political leaders, the stakeholders, their constituents, and the general public. To bridge communication gaps, many decision-makers without formal scientific training have become very knowledgeable about science and its limitations. Conversely, many scientists have become more conversant in the decision-making process

Problems arise when decision-makers are not adequately informed, for whatever reason, or their expectations are not met. Under these circumstances they may relegate scientific findings to a less integral position in the decision-making process and choose an action or alternative that is the most politically expedient. Upon viewing decisions of this kind, scientists often believe that their research has had little effect on the decision. They sometimes believe that the only research that gets used in decision-making is research that supports a decision that has already been made, implicitly or explicitly.

To complicate things further, scientists often believe that scientific findings should be preeminent in the decision-making process and so do not fully appreciate the decision-maker's need to consider the societal implications of the decision at hand. Sometimes in an effort to make their work more relevant to the decision-making process, or through pressure from the decision-maker or interest groups, scientists propose alternatives that are not their preferred actions, but ones they believe would be politically acceptable, thereby perpetuating the system of behaviors that leads to politically expedient decisions.

Scientific disagreement and conflict over appropriate recovery strategies also can lead to politically expedient decisions. Disagreement can stem from differences in the problems being addressed by different analyses, differences in analytical approaches to the problems, differences in the assumptions underlying each of the approaches, the quantity and quality of the available data, and the rigor with which each approach deals with the complex life cycles and habitats of the species. Scientific debates are to be expected; they are a part of the scientific process and are necessary for winnowing through the various scientific approaches to expose their weaknesses and strengths. These debates can become preeminent for the involved scientists, but some of the decision-makers we spoke with found them confusing and are more interested in identifying broad-scale patterns or areas where the various approaches agree.

Unfortunately, decision-makers can use scientific uncertainty to justify politically expedient decisions or to defer decisions pending further research, a strategy that is often believed by both scientists and decision-makers to reduce uncertainty and narrow the scope of the decision. An example of a deferred decision was NMFS' decision to delay breaching the four lower Snake River dams. When decisions are deferred for reasons not obvious or not defensible to scientists, who believe they have sufficient scientific support to justify a decision, they again feel their research is being ignored in favor of politics. This is the way many scientists feel about the decision to delay breaching.

Individual Models and the Decision Process

Most of the analytical frameworks that the ISAB reviewed were developed to address the specific needs of an agency. In these cases the model probably will be of greatest utility to the sponsoring agency. Two of the modeling efforts we reviewed seemed to enjoy very successful communication with their respective decision-makers. These were ICBEMP-BBN and CRI. In both cases, the decision-making authority was part of the same organization that had ownership in the model and scientists developing the models belong to the organization. In both cases, the decision questions that the modeling effort was intended to inform were well-defined in advance of the modeling project, the models were directed at those questions, and the decision-makers seemed not to experience difficulty in understanding and using the model results. It is noteworthy that in those two cases, the treatment of uncertainty in the models did not seem to pose a special obstacle to the decision-maker who was going to use the results.

The EDT model was developed to serve two rather distinct purposes: (1) to articulate a particular desired vision for the future of the basin with a prescription for associated management strategies, and (2) to appraise the expected effectiveness of specific proposed projects in the context of the watersheds where they would be located. The strategic vision application is not tightly linked to an identified decision process. The decision process of project funding by the Fish and Wildlife Program intends to incorporate the project proposal application, but that role has not yet been formalized or implemented.

The CRITFC model has yet to be used for decision-making, and its results have not yet been promoted as arguments for influencing a particular decision. The history of PATH in relation to decision making processes, however, is complicated enough to warrant the following separate section.

<u>The Institutional History of PATH</u>

It is difficult to characterize PATH accurately and fairly, because it was never just one thing; many things happened under the PATH umbrella. It is important, however, to understand PATH accurately and fairly for two reasons: 1) because PATH did attempt some worthwhile things that have not yet been duplicated by the other modeling efforts, and 2) because feelings have run strong on several aspects of the modeling debate, and some of the PATH participants still function as a distinct constituency (e.g., PATH-EM, this report).

PATH was a "process" that was initiated as part of the settlement of the litigation over the 1995 Hydro BiOp. At that time (1995) there were two dueling models, FLUSH and CRiSP. FLUSH was the work of a coalition of states and tribes and put a heavy emphasis on the effects of flow on smolt survival during downstream passage. CRiSP was the work of Dr. Jim Anderson's group at the University of Washington, funded by BPA, and it emphasized other factors affecting smolt survival during downstream passage. NMFS, the agency responsible for the BiOp under ESA, did not have ownership of either model, and at the time of the 1995 Biological Opinion, NMFS basically declined to decide between them. The legal resolution, at that time, was to set up PATH, with representation of all the main parties to the debate, as a scientific forum that would be given the assignment to achieve scientific consensus in time for the next BiOp on hydro operations, which was then scheduled for 1999.

Though NMFS had representation in PATH, NMFS did not have administrative responsibility or control over PATH. The bulk of PATH funding came from BPA through the Council’s Fish and Wildlife Program, and some of the PATH funding came via other BPA mechanisms. PATH developed its own organizational structure with the engagement of a consulting company (ESSA) that provided a guiding facilitator and with the establishment of a Scientific Review Panel of distinguished fisheries scientists. Later, as the technical strategy for the PATH analysis took shape, PATH also engaged some additional fisheries modeling expertise, some of it from outside the Columbia Basin, to actually carry out the central mathematical analysis.

PATH then set out on four separate, but related enterprises. These were:

1. continuing work on the FLUSH and CRiSP models, more or less independently of one another and not calibrated to the same data sets,

2. continuing work to consolidate a common database of "run reconstructions" for a list of "index stocks,"

3. construction of a new hypermodel that would incorporate both FLUSH and CRiSP as modules for calculating downstream passage smolt mortality and that would fit all its other parameters via Bayesian inference, calibrating to the "run reconstructions" of "index stocks,"

4. construction of a decision theoretic framework for incorporating the expert judgement of the scientific review panel into a set of recommendations that would respond to the accumulated evidence (including the results from the hypermodel), the uncertainty, and some appraisal of the risks.

These four pieces each made sense in their own way, but they did not fit together satisfactorily. Superficially, one would think that the priority charge to PATH was to diagnose and reconcile the differences between FLUSH and CRiSP, but, taken together, components (1) and (3) of the PATH enterprise largely precluded that resolution. The technical Bayesian analysis in component (3), while genuinely leading edge in its sophistication, was very complicated and difficult to explain to potential customers for PATH's results. Component (4) edged into the business of making recommendations about hydro operations and the merits of dam breaching, which may have been perceived in some quarters as going beyond the original PATH charter.

Notwithstanding a large amount of good work, PATH was vulnerable to the criticism that it had not answered the question asked of it, and that it was trying to answer policy-sensitive questions that had not been asked of it. It was not quite that simple, of course, because it was not entirely clear who was in charge of PATH, or to whom PATH was supposed to answer. Furthermore, because of its "consensus" structure, PATH was notoriously slow in operation, and it generated unwieldy volumes of reports.

The PATH funding was piecemeal. Many of the subprojects of PATH were presented annually to the Fish and Wildlife Program as separate proposals for funding. In 1999, the ISRP recommended that the PATH proposals not be funded. Subsequently, the Northwest Power Planning Council, which makes the final decisions on program funding, decided to terminate funding for PATH as an organization.

For the past year, NMFS has been doing its own modeling and data analysis to support its own decision processes under its ESA responsibilities. The NMFS approach, so far, has been simpler and more statistically consistently than was PATH.

CRiSP development continues in Dr. Jim Anderson's group. Elements of FLUSH are maintained by some of the players in the coalition of states and tribes. This coalition has formalized itself as an organization called STUFA, but with CRITFC spinning off yet another modeling effort that resembles EDT and ICBEMP-BBN in its use of a database of habitat characteristics and with the overlay of a conventional treatment of population dynamics.

The review functions of PATH's scientific review panel, the facilitation function of ESSA, the hypermodel effort, and the consensus function that PATH as a whole was supposed to accomplish, no longer occur in that forum. Nevertheless, many of the individuals who were involved in PATH continue to work on Columbia Basin salmon modeling and data analysis issues--and some results of the PATH activities are continuing to move into the peer reviewed scientific literature. The habitat-centered modeling efforts of ICBEMP-BBN and EDT have taken up important aspects of salmon modeling with which PATH, for the most part, was not concerned.

The final product from PATH as a functioning organization was the report on Experimental Management, PATH-EM. This report constituted a continuation of PATH’s commitment to evaluate the time necessary to detect effects of anticipated management actions and to provide some preliminary mathematical estimates of the effects, though it may have been a somewhat rushed and incomplete effort owing to the termination of PATH’s funding.

The logic of decision analysis often leads to a conclusion that when relevant uncertainties are large, it is imprudent to commit to long-term courses of action unless these courses of action have a built-in flexibility to respond to new information. In fact, the decision options to be weighed include the relative investment in continued data collection versus the merits of experimental manipulations. The relative merits of the two options should be judged in terms of the value of the information they provide, quite aside from the direct outcomes any experimental manipulation expressed in terms of the desired objective (such as more fish available for harvest, or an increased probability of population persistence). The value of new information is quantified in terms of its potential use to influence the selection of subsequent courses of action. The formalization of such flexible decision rules together with the cost-benefit of continued data and experimental manipulation are encapsulated in the technical theory of Adaptive Management.

Among the Columbia River basin salmon modeling efforts, the PATH experimental management report represents the most substantive attempt, to date, to provide an analysis of the prospects for actual Adaptive Management. The analysis described specific experiments and provided a preliminary assessment of the types of information that the experiments would provide, and how much time they would require for completion. Because there seems to be a temptation, in many quarters, to co-opt the term “adaptive management” merely to put a positive spin on vague management plans, it is important that this kind of technical analysis of adaptive management resume as an ongoing enterprise connected to actual decision making in the Basin.

Communication and Sources of Credibility

The best resolution of scientific disagreement is through publication in the open scientific literature, and provision of access to model code and to data files allowing independent verification. Data access should include access to the original primary data and metadata. If there is not access to the primary data, derived quantities, when treated as if they were data, may carry error that escapes scrutiny. Because publication imposes time delays that may be inconsistent with decision timetables, it is advantageous for a modeling project to have good lines of communication with decision-makers. If a modeling effort is motivated by a desire to contribute to a particular decision, it is especially helpful to invest up front in enough communication to ensure that the model really is addressing the right question.

The present culture of Columbia Basin salmon science has not put a great premium on publication. This greatly magnifies the importance of the decision-makers' *a priori* trust in any particular modeling effort that they choose to use. This is not entirely a desirable situation. A greater role of publication in the open literature in establishing credibility for the use of science in Columbia Basin decision-making would be good. A greater reliance on the mechanisms of normal scientific discourse might also reduce some of the contentiousness that has characterized the history of scientific debate over key issues in Columbia Basin salmon management.

We might note that the recent experience of an attempt at a scientific consensus "process" operating within the conventional Columbia Basin culture was not especially encouraging. The process proved to be very slow and cumbersome--so much so that it probably was no quicker than publication might have been. Furthermore, the consensus did not hold.

Realistically, it is inevitable that institutional trust will give selected models and modelers an inside track for access to decision-makers, but a culture of publication and respect for published results could still influence the standards of quality control and the habits of discourse for resolving scientific disagreement. We view as a very positive development the recent example of CRI in espousing a culture of publication and a stance of healthy skepticism about traditional data sources.

Appendix I: Technicalities of Quantification of Uncertainty

Theory can usefully distinguish three main kinds of "error" as contributing to the uncertainty of a model's predictions. These are measurement error, parameter error, and process error. The label "error" in this theoretical language is meant to refer to a source of variance, not to a "mistake." The role of variance in our characterization of this type of error is crucial. We quantify the error in terms of a variance, because we acknowledge that we cannot know the actual value of the error in a given instance, but only can know its probability distribution. If we knew the actual value of the error, we would just subtract that from our estimate, and then there would not be any error.

Measurement error refers to the random influences that cause an observation to deviate from the true value that is being measured. For example, we may obtain observations in the form of redd counts in an attempt to census the size of a spawning population. However, because this is not an exhaustive count of the population, but rather a count in an index area, during a particular time window, which then is expanded to estimate the entire spawning population for that year, the estimate of population size has an error component.

Parameter error refers to random influences that cause a parameter estimate to deviate from the true value that we wish it to represent. Use of measurements that are subject to measurement error can be one contributor to parameter error. Another important source of parameter error that can arise even with no measurement error is sampling variation. Consider, for example, that our parameter of interest is the average number of recruits per spawner at a particular population size. We know that the actual number of recruits per spawner in a given generation is subject to random environmental influences. If our data set consists of only a few years' observations of numbers of recruits for brood years when the spawner population was in a particular size range, that small sample may misrepresent the true average, even if the measurements themselves had no measurement error. In reality, for a quantity like average number of recruits per spawner at a particular population size, we would expect the parameter error to be substantially influenced both by measurement error and sampling variation.

A further contributor to parameter error is simple mis-specification of a statistical extrapolation procedure. Consider for example, that the parameter of interest is the alpha of a Ricker production equation. That parameter represents the limiting value of recruits per spawner in the limiting condition of a vanishingly sparse population. Since our data come from populations that are not (yet) vanishingly sparse, we need somehow to extrapolate from the estimates of recruits per spawner at observed densities to an estimate of what the value should be at the limiting density where the parameter is thought to express itself in pure form. In the case of the Ricker equation, we do this by linear regression, since the equation represents the relation between spawner density and recruits per spawner as a linear relation, in which the parameter alpha is the y-intercept. This linear regression seems a clever work around for the absence of actual data at zero density, but what if the true relationship between spawner density and average recruits per spawner is not exactly linear? Then the linear regression will have introduced an error component over and above the sampling variation in the brood years for which we have observations, and the measurement error in those observations. We see, therefore, that correctly characterizing the parameter error is a rather complicated undertaking.

Finally, when we apply a model to make predictions of a process that itself is subject to random variation, called process error, the process error will contribute to the variance in the distribution of the discrepancy between the prediction and what actually will occur. The influence of process error -- the inherent randomness in the process -- will cause prediction error even if we knew the parameters exactly and our measurements were perfect. Consider the case of predicting the number of recruits, from knowledge of the number of spawners that give rise to that brood and from knowledge of the parameters of the Ricker equation that applied to that stock. The Ricker equation gives the "average" number of recruits per spawner that we should expect from a given brood. Even if we knew the Ricker parameters exactly, the prediction is only for the "average." On the other hand, the actual number that will return in the year being predicted will be influenced by random environmental variation that in principle is not predictable, because it is future noise in the process.

While we may not be able to fully predict the deviation owing to the random process error of this sort for a given future set of returns, we should be able characterize its distribution. That is, we should, in this example, be able to say that the number of recruits will be a distribution, more or less centered over the predicted average from use of the Ricker equation applied to our estimate of the number of spawners that gave rise to this brood, and using the parameter estimates that we obtained from fitting the Ricker equation to some historical data. This distribution of the predicted returns will also have a spread, measured usually by its standard deviation, that reflects the uncertainty in the prediction. This prediction uncertainty will have contributions from measurement error in the estimate of the starting number of spawners, parameter error owing both to sampling variation in the brood years from which data were available to estimate the parameters of the Ricker equation, and owing also to measurement error in those data, and process error in the actual mechanisms that influence the recruits per spawner for a given brood.

Because this error exists, any manager who intends to make decisions based on the prediction needs to know it and take it into account. In this stock recruitment example we expect the total prediction error to be considerable, based on our experience from the substantial scatter that we see in the fit of observations to empirical stock recruitment curves. This scatter is a combined result of measurement error, parameter error, and process error. The first practical question that arises in this connection is: can we adequately estimate the total prediction error just from the scatter in these empirical plots?

The answer, unfortunately, is no. The scatter in the plot has contributions from all three sources of error, and it represents a distribution of the discrepancy between measurements and prediction -- not between prediction and truth. And the measurements are subject to measurement error, and the predictions are subject to measurement error, parameter error and process error. If we use the discrepancy between measurements and prediction in the scatter plot as a quantification of the prediction error, we will have overestimated the short-term prediction error. The misrepresentation of the long-term prediction error will likely be even larger and, for rather complicated reasons, we will not even know whether the long-term prediction error will have been over estimated or under estimated.

Bear in mind that our estimation procedure for this stock recruitment relationship example was based entirely on a short-term relationship, namely that between spawners and recruits over the span of one generation. If we then use this to make a longer-term prediction, over the span of several generations, the three error sources (measurement, parameter, and process) will propagate differently. Note that a modeling application such as a calculation of the probability of extinction has this character of projecting for several generations into the future.

Measurement error for initial conditions (basically the estimate of the number of spawners for this brood, in this example) enters the projection once, as a simple multiplicative factor. Parameter error will be a fixed value that holds for the duration of the projection, but its effect will compound, more or less exponentially, with each generation. Process error gets re-sampled each generation, because a new expression of the environmental "noise" manifests itself, so there will be a degree to which the process error "averages out" over the generations, even though its effect compounds in each generation. Thus we will get quite different results for the prediction and for the prediction error of the projection, depending on how we apportion the error that we see in the scatter plot among measurement error, parameter error, and process error.

As a result, it is not so simple a matter as automatically assigning all the regression scatter to the process error (which is what the parameter estimation step of the Dennis model does), or automatically equating the jack-knife cross-validation residual sum of squares with prediction error (which is what would happen in a naive comparison of observed versus predicted conditions in a watershed with the EDT model). The correct calculation of measurement error, parameter error, and process error is a technically involved undertaking that is not simply superimposed on a modeling effort as an afterthought. It has to be built into the modeling, and it is a demanding part of the project.

The technical demands escalate as the model becomes more complex. For example, fairly simple traditional methods for putting a conventional "confidence interval" on the estimate of a single parameter may not go too far astray for error analysis purposes. As soon as there is more than one parameter involved, however, a whole world of ambiguities open up if we try to estimate the prediction error by simply propagating all the "confidence distributions" independently for all the parameters.

First of all, the parameter errors are generally not independent; and secondly, conventional confidence distributions do not have the right probability properties for use in error propagation. This is reasonably well known in the technical risk assessment literature, but the solution involves considerable sophistication, and there is not a clear consensus in this literature on all the details of implementing the solution. Broadly speaking, the key elements of the solution involve Bayesian methods to compute a "joint posterior distribution" on the entire set of relevant parameters. The joint distribution takes account of the correlation in error between various parameters, and usually results in a much smaller computed value for the total prediction error than would result from independently propagating all the confidence distributions on the parameters. Formally, a posterior distribution has the right mathematical properties for combination with other probabilities.

Difficulties with propagating parameter error are revealed clearly enough in simple applications of the Dennis model for estimating distributions of time to extinction. The forward projection stage of the Dennis model is a Brownian motion model, which represents process error in its diffusion term. In the parameter estimation stage of the Dennis model, confidence limits are put on the "average growth rate" parameter, because this falls out directly from the regression machinery that estimates this parameter as a slope. This average growth rate parameter becomes the drift term in the Brownian motion forward projection. Unfortunately no confidence interval is put on the estimate of the "variance in growth rate" which becomes the diffusion term in the Brownian motion projection, even though the uncertainty in this parameter is bound to be large (the sampling variance for a variance is notoriously large, and declines slowly as the sample size increases), and this parameter has a large influence on the result.

Furthermore, it is not really clear in the Dennis model framework how to propagate the confidence calculation for the one parameter, the drift term that is given a confidence interval, through the prediction of extinction times in the projection stage. Simple expedients, such as separately calculating distributions of extinction times using the upper 95% confidence limit, the central estimate (MLE), and the lower 95% confidence limit of the average growth rate parameter create new problems. What is the meaning, for example, of the lower 15% tail of the time to extinction using the upper 95% confidence limit of the average growth rate parameter? It is not mathematically legitimate to fold the "confidence" from a parameter estimate into the probability of outcome in a projection conditioned on a particular data set. The two quantities are of fundamentally different kinds -- too different to be combined.

The concession to simplicity of omitting density dependence in the Dennis model saves it the difficulty of having to deal with even more parameters that would be even more uncertain. The omission of density dependence, however, leads to other serious problems. Part of the reason that conventional stock recruitment analysis has not revealed strong density dependence in the Snake River chinook is, almost certainly, because the carrying capacity of this system has not remained constant during the interval spanned by the data. The dominant dynamics for the past several decades has been of populations tracking the downward spiral of their carrying capacity in a way that is "density dependent," but does not give a pretty Ricker-curve picture if we try to analyze it, incorrectly, with an analysis that assumes constancy of the density dependence parameters for the duration of the data set.

These comments on the treatment of uncertainty analysis in simple applications of the Dennis model do not detract from the many very positive aspects of CRI. In addition, for the most immediate uses of this chapter of CRI in decision-making, these criticisms are probably irrelevant. The immediate use of the Dennis model in CRI has been to determine whether the various stocks are in imminent risk of extinction, and the answer has already come in that they are. Building in more uncertainty analysis, just to show that the risk of extinction perhaps is even higher, is probably unnecessary. Where the uncertainty analysis will be important to the actual decision-making will be in the detailed analyses to make selections among various interventions.

Appendix II. Description of Decision Support Systems Used to Guide Salmon Recovery Planning in the Columbia River Basin

This appendix contains descriptions of different decision support tools (models) currently informing salmon policy in the Columbia River Basin. We include descriptions of four such quantitative analytical systems: Ecosystem Diagnosis and Treatment (EDT), Cumulative Risk Initiative (CRI), Bayesian Belief Networks (BBN) of the Interior Columbia Basin Ecosystem Management Project, and Plan for Analyzing and Testing Hypotheses (PATH). The descriptions were taken with minor editing directly from reports by primary authors of each modeling effort and are therefore in their own words. Lars Mobrand of Mobrand Biometrics, Inc. described EDT, Peter Kareiva of the NMFS Northwest Fisheries Science Center described CRI, Bruce Rieman of the USDA Forest Service Intermountain Research Station described BBN, and David Marmorek of ESSA Technology, Ltd. described PATH.

The following summaries do not contain all of the information needed to understand the analytical systems in detail. Readers are referred to appropriate publications by project authors for more complete descriptions.

Ecosystem Diagnosis and Treatment (EDT)

<u>Steps in the EDT analysis</u>

The basic steps of EDT are captured in its name, Ecosystem Diagnosis and Treatment:

> <u>Ecosystem</u>: The description of the biophysical environment of species and populations of concern.
> <u>Diagnosis</u>: Evaluation of the "health" or quality of the environment with respect to specific species or populations.
> <u>Treatment</u>: The analysis of the impact of different strategies in changing the existing environment toward one that is more compatible with the needs of the species or population of interest.

More explicitly, in EDT we:

1. Describe the habitat template in terms of a set of physical attributes of the current terrestrial and aquatic environment at the level of the HUC-6 (hydrologic unit code) .

2. Assess the habitat template in regard to how it affects biological performance measured as productivity, capacity and life history diversity of salmon and other species – i.e. derive a "survival landscape" based on habitat conditions.

3. Evaluate how regional alternatives might change biological performance by relating strategies to changes in environmental attributes.

4. Capture our accumulated knowledge about the relationship between salmonids and other aquatic and terrestrial species and their environment in the form of documented "rules" that explain or hypothesize survival responses to their habitat – i.e. organize a repository of knowledge and information.

5. Perform sensitivity analysis to guide refinement and modification of management strategies and or objectives and development of monitoring and evaluation plans.

More details on EDT and the Framework Project can be found at the Framework web site: http:/www.nwframework.com.

The data used in EDT

There are four major types of data used in EDT. The first data category consists of the 108 strategies that are selected and combined to make up an alternative. In the Framework Project, these are arrayed across 10 ecological provinces to make an alternative focused on a particular vision.

In EDT, we distinguish information that is actually observed from information that is derived from other information. Most of the information that is routinely used in natural resource management is actually derived from a smaller set of real observations. For example, counts of adult salmon at mainstem dams, abundance estimates of spawning fish and the number of fish harvested are all basic fisheries information that is expanded or derived from a much smaller set of actual observations. As you might expect, to describe the habitat of the entire Columbia River, we have to derive a lot of the information. We note where these derivations occur and base them on a set of explicit rules. The description begins by filling in the available information for each of the 7,200 HUC-6 units. Where information is missing, the scientific literature and appropriate experts are consulted to derive rules for filling in the gaps.

The next higher level of derived information is the assessment of the habitat with respect to specific fish or wildlife species. Habitat quality is assessed for each life stage in terms of productivity and capacity. For chinook salmon we distinguish habitat quality for each of 16 life stages.

Finally, the productivity and capacity is integrated over the entire life history to derive the overall estimated productivity and capacity. A number of life history pathways are tested to assess the impact on life history diversity as well.

The different data types are related through a set of rules that are the heart of the EDT expert system. They are the basis for describing habitat and relating habitat observations to the higher level environmental attributes and the resulting biological response. The rules capture the region's expertise and are derived from empirical research, the scientific literature and expert opinion.

There are four types of rules that call on different types of expertise. The first links each of the 108 strategies to one or more of 51 types of habitat observations. Those devising the strategies and alternatives intend them to result in some change in the environment. This is formalized by capturing the knowledge of how different types of actions (strategies) affect our 51 descriptors of habitat conditions.

The second type of rule links the observations to 44 Environmental Attributes. These rules describe how the habitat observations are expanded to account for incomplete or missing information. These rules are devised by hydrologists, geomorphologists and other physical scientists to expand the direct habitat observational data to all 7,200 HUC-6 units for all time periods.

The third rule category is based on the knowledge of biologists regarding life stage survival response or productivity of specific species to one or more of the Environmental Attributes for each HUC-6. The resulting Biological Metrics can be thought of as 19 graphs showing the relationship between life stage productivity and Environmental Attributes.

The final type of rule is an algorithm that integrates over the entire life history pathway to compute total productivity and capacity. This is done for each successful life history pathway to estimate life history diversity.

The scale and resolution of the EDT analysis

EDT paints a picture of the biological and physical landscape of the Columbia River. The “pixel” size of this picture is the HUC-6 (hydrologic unit code) of which there are approximately 7,200 in the Columbia River basin. These are organized in a spatial hierarchy to describe subbasin, ecological provinces and the Columbia River basin. EDT describes the equilibrium condition of the basin as a result of a set of strategies. Time is not a factor in EDT except in regard to the explicit description of the various life history pathways of target species.

Biological resolution in EDT is limited only by the available data. The present analysis assesses habitat conditions in terms of four species: two aquatic species, chinook salmon and bull trout, and two terrestrial species, black bear and beaver. Data has been assembled to distinguish 107 natural and 50 hatchery populations of chinook salmon as well.

Measures of “performance” in the EDT analysis

EDT measures biological performance in terms of three population parameters: productivity, abundance potential, and life history diversity. Productivity is the density independent component of survival times the rate of reproduction. Abundance potential is the carrying capacity of the habitat. Productivity and abundance potential are assessed for habitat in each life history stage. They are integrated over the life history to describe performance of a particular life history pathway. The grain and variation of the habitat description means that there can be more than one potentially successful pathway across the "survival landscape." This provides an assessment of the strategies in terms of their positive or negative effect on life history diversity as well as population productivity and abundance. All three parameters reflect the ability of the *environment* to support salmon production *in the long term*.

General philosophy and aim of the EDT

The general philosophy of EDT is well grounded in established scientific principles describing the influence of habitat structure on the characteristics of biological communities. Specifically, EDT assumes that habitat characteristics determine biological performance. Hence, changing habitat through natural events or by human action will have a corresponding effect on biological performance.

A second philosophical point is that EDT recognizes that the scientific basis for decisions regarding the future of the Columbia River cannot and will not be limited to statistically "proven" knowledge. While statistically based information is important, scientists are increasingly aware that the complex and dynamic nature of ecological systems is currently, and perhaps always will be, imperfectly captured in statistical relationships. Prudent management must take advantage of all available information. For this reason, EDT uses both statistically based and heuristic knowledge.

The current application of EDT is to the Multi-Species Framework Project sponsored by the Northwest Power Planning Council and federal and tribal management agencies. The focus of the Framework Project is on the long-term vision for fish and wildlife management in the Columbia River. Development of a vision involves consideration of the types of strategies required and their impacts in terms of human communities and other social and economic factors in addition to their ecological impacts. EDT is being used in this context as a tool to facilitate long-term basin-wide planning. However, it also will provide a basis for development of more detailed and shorter term plans for individual subbasins.

Cumulative Risk Assessment (CRI)

The four key steps to a CRI analysis

1.) Estimate the risk of quasi-extinction for known populations
2.) Construct demographic projection matrices that depict current demographic performance rates and in turn can be used to calculate annual population growth rates (assuming a "current conditions").
3.) Perform sensitivity analyses to assess where in the life cycles of salmonids there are the greatest opportunities for promoting recovery, as measured by changes in the annual population growth rate. This can be done several different ways. The simplest is to manipulate the values in baseline matrices to represent particular demographic improvements, and calculate the % increase in annual population growth rate that

results. This increase in annual population growth can then be converted into an estimated reduction in quasi-extinction risk.

4.) For those demographic improvements that give a noteworthy response in terms of population growth, identify management actions that might accomplish those improvements, and use statistical analyses or experimental studies to determine whether there is evidence that those improvements are actually feasible with the management action being considered.

More details can be found in several other documents available from the CRI website (http:/www.nwfsc.noaa.gov/cri/).

The data used in CRI

The primary data used by CRI are time series of population counts, and recruits per spawner ratios. An example is given below, from fall chinook salmon in the Snake River.

Table 1. Counts for Fall chinook salmon.

year	spawners	Recruits to spawning grounds (total)	Recruits to spawning grounds (minus jacks)
1980	515	2294	1285
1981	878	1555	983
1982	1209	1810	1224
1983	909	1986	1115
1984	717	1764	934
1985	1080	654	541
1986	1403	706	539
1987	1064	373	292
1988	702	747	710
1989	815	656	529
1990	273	284	227
1991	767	300	206
1992	674		
1993	883		
1994	448		
1995	226		
1996	964		

From this, one can calculate an extinction risk, and estimate how much we need to increase annual population growth to mitigate this risk, as shown:

Table 2. Quasi-extinction risks for Snake River fall chinook salmon (based on data from 1980-1996).

	Avg. λ	avg. N over last 5 years	p(one spawner within 10 yrs)	p(one spawner within 100 yrs)
Fall Chinook	1.13 (0.89-1.44)	639	<0.0001 (<0.0001-0.16)	0.06 (0.0002-1.0)

Table 3. Quasi-extinction probability for Snake River Fall Chinook associated with particular increases in λ.

% change in λ	p(one spawner within 100 years) Fall Chinook
5	0.005
10	0.0003
15	1.3×10^{-5}
20	7.1×10^{-7}

It may also be possible to construct a detailed demographic matrix that can then be used to simulate management experiments such as harvest reductions. Below, as an example is the Snake River fall chinook salmon demographic matrix.

	2	3	4	5	6
Age frequency of females (f_x)	0	0.129	0.652	0.198	0.020
93-96 Ocean harvest rate (h_x)	0.0123	0.0465	0.1368	0.1838	0.1953
Female eggs per female spawner (m_x)		1442.5	1566.5	1625.5	1625.5
Propensity to breed (b_x)	0	0.081	0.648	0.859	1.0

93-96 Mainstem adult harvest rate	0.174
93-96 adult Bon to Basin conversion rate	0.471
s_1	0.0102

These parameters are then substituted into the following matrix where, μ represents the age-specific fraction of ocean dwelling salmon that return to spawn (it combine the probability of returning during that year, with the survival rate swimming upstream, which includes harvest reductions as well as other mortality), s_1 represents survival during the first year of life, s_A is survival as adults living in the ocean, and h_i indicates ocean harvest rates on fish in ageclass i.

	1	2	3	4	5	6
1	0	0	$(1-\mu)s_1b_3m_3$	$(1-\mu)s_1b_4m_4$	$(1-\mu)s_1m_5$	$(1-\mu)s_1m_6$
2	$(1-h_2)s_A$	0	0	0	0	0
3	0	$(1-h_3)\, s_A$	0	0	0	0
4	0	0	$(1-b_3)(1-h_4)s_A$	0	0	0
5	0	0	0	$(1-b_4)(1-h_5)s_A$	0	0
6	0	0	0	0	$(1-b_5)(1-h_6)s_A$	0

This matrix can be used to simulate the consequences of reduced harvest, and other management actions. Importantly, for many management actions (almost everything other than harvest reductions) it is not certain whether a given action will accomplish the desired demographic improvement. This is where the "feasibility studies" come into play. For a feasibility study, the dependent variable will typically be recruits per spawner, number of spawners, smolts per spawner, smolt-to-adult returns, or survival during some life stage. Correlations are then sought between these measures of salmonid productivity and variables such as number of hatchery releases, fraction of stream miles failing to meet EPA water quality standards, and so forth. What CRI envisions as feasibility studies also represent evidence that EDT uses in constructing its "rules" relating stream attributes to salmon production.

The scale and resolution of CRI analyses

CRI is most effective when applied to distinct populations, or collections of populations. This is because it focuses on population growth rate and a population's risk of extinction. The spatial scale at which CRI best operates ranges from subwatershed on up to subbasin or basin. As it is currently developed, CRI is not equipped to deal with an entire province or region comprised of many populations and multiple ESUs. However, there are plans for extending the CRI to this large scale (beginning with a technical workshop in December 2000 at NWFSC, which was aimed at multiple populations and ESU-wide priority-setting). CRI would never be used at the fine scale of a particular reach or stream. CRI could never inform us about reach-specific or small-scale management actions. The output of CRI often takes the form of: "*if this, then the expected response is ___*". CRI does not deal with individual fish at all, and also does not deal with life history diversity. In the absence of data and statistical relationships, the CRI does not venture very far with its analysis.

Measures of "performance" for the CRI analysis

The primary measure of performance for CRI is average annual rate of population growth. This core measure is then the basis for two additional measures of performance: risk of extinction over 10 years and 100 years, and the percentage by which annual population growth is expected to increase with some management action. Although it is impossible to validate "risk of extinction" as a performance measure, annual population growth rate and % change in annual population growth can be validated – these are both measurable, and in fact are routinely available from the type of spawner or redd counts typically made for salmonids.

<u>CRI's general philosophy and aims</u>

CRI's three most distinctive features are:

1.) an emphasis on simplicity and simple models, so that others outside NMFS can repeat their own analyses with slight modifications of the assumptions, new data, different time periods, different levels of risk averseness, and so forth,

2.) a staunch empiricist's skepticism, such that a priority is placed on relationships supported by data, and that otherwise must be couched as "*if this, then that*" statements,

3.) focusing on population dynamics or demography as the window through which to evaluate management actions.

Bayesian Belief Networks (BBN) of the Interior Columbia Basin Ecosystem Management Project

We were asked to evaluate the potential effects of three land management alternatives on fishes and their habitats across the entire Interior Columbia River Basin (ICRB). The ICRB represents an area of about 58 million hectares, and roughly 7,000 6^{th}-code watersheds (also known as subwatersheds) (Figure 1) supporting 88 taxa of native fishes. Those alternatives included spatially explicit aquatic conservation and restoration strategies that dictated such things as riparian buffers, watershed analysis, and a system of designated watersheds where aquatic conservation was emphasized over all other management objectives. Information available for this analysis included an existing, probabilistic assessment of the current status and distribution of seven "key salmonid" fishes. It also included independent, spatially explicit projections of the intensity and distribution of ground disturbing/mitigating activities (e.g. logging, road building, road closure) and anticipated changes in large scale patterns of wild fire associated with each alternative.

Any evaluation of land management effects for the entire ICRB is a daunting task. It is obviously complicated by the sheer size and complexity of the basin, and by the relative lack of information and understanding of the physical and ecological processes relevant at such a large scale. This analysis was further complicated by an urgency in the decision process that dictated an analysis completed in the course of only a few months.

There have been two basic approaches to the evaluation of broad scale land management proposals. One, characterized in earlier iterations of the ICRB planning process and other large Federal Assessments has used a summary of expert opinion. In essence, available information is arrayed before a panel of scientists or resource management specialists who provide an interpretation of likely outcomes based on their background and experience. Although simulation or empirical models for some specific linkages or environments might represent some of the available information, the system is characterized largely in the participants minds. This approach proved unsatisfying for several reasons: (1) the huge body of information made it difficult to conceptualize and account for multiple interacting effects; (2) The influence of the experts' assumptions on results of analyses could not be evaluated directly; (3) the analyses could not be revisited or updated easily when management drivers or key assumptions were modified; and (4) the results were not easily quantified or made spatially explicit and thus could not provide a clear contrast among alternatives.

A second approach has been model based where predictions of species/system responses are made through mathematical abstractions that attempt to characterize the relevant physical and ecological processes. Formal, model based approaches directly address the last three issues linked to expert judgement above. Important issues with model based approaches include the invariable lack of information needed to parameterize, or the computational resources and time to apply, very detailed models for more than a handful of landscapes or populations. Although attempts to model whole ecosystems have often incorporated remarkable complexity (e.g. EDT), it is not clear that mechanistic detail necessarily leads to better predictions.

In our evaluation we attempted to resolve these issues by combining expert judgement and a relatively simple, but formal model framework of Bayesian belief networks (BBNs). Also known as Influence Diagrams, Causal Trees, or Probability Networks, these networks are typically represented by box and arrow diagrams that depict causal linkages among the physical and ecological factors that influence the probability of an outcome or state of interest (e.g. the status of a bull trout population). The linkages between elements or “nodes” of the diagram are represented by conditional probability tables that reflect the probability or frequency of a particular state given the states in all “parent” nodes that directly influence the one of interest.

Bayesian Belief Networks provided an appealing framework for the ICBRB analysis for several reasons. First, the networks are explicit, formal representations of our knowledge, beliefs, and uncertainty for complex ecological systems. Essentially these networks provide a means of articulating what we think about the causal web influencing species and their habitats. The analysis and the logic underpinning it become far more transparent than one based solely on expert judgment.

Second, representation of the linkages and outcomes as probabilities acknowledges the lack of precision in predictive models that may arise through limited understanding of the system, inherent variation in biological and physical processes, and lack of data. A distinct advantage of this approach is the ability to combine information derived empirically and from expert opinion. The subjective element of “expert” judgment is clearly a contentious point, but in reality for this kind of analysis there is no other option. By incorporating information from multiple advisors, areas of consensus and disparity of opinion become obvious and can be reflected directly in the conditional probabilities. Where a particular relationship is understood well experts and data will tend to converge on a strong distinction in probabilities among alternative states. Where it is not, differences of opinion or noise in the data will produce a more uniform distribution among states reflecting either the lack of a strong ecological linkage or a lack of understanding.

Third, the networks can be as simple or complex as information and understanding allow. The absence of mechanistic detail does not necessarily limit the utility of a model. For example there is good evidence of a relationship between the level of land disturbance and the occurrence and persistence of bull trout. Most experts would agree that substantial watershed disruption associated with high densities of roads and logging will increase the likelihood that a bull trout population will not be strong. A simple and potentially useful model might include only that single linkage. Alternatively, a model could include additional detail such as the expected links between canopy removal, stream temperature, juvenile growth and survival, population growth rate, and extinction risk. The additional detail may be more appealing, but far more difficult to support for an analysis across a huge region. The construction of networks is an exercise in compromise that attempts to balance mechanistic detail with the strength of the relationship implied by any linkage and the information and understanding available to drive the model. Collapsing the model to fewer, simpler linkages does not necessarily compromise the ability to make a prediction if the fundamental tie between the ultimate driver (amount of land disturbance) and outcome of interest (bull trout status) is believed to be relatively strong.

Finally, the computational demands are relatively simple. Networks can be constructed easily with commercially available software or traditional statistical packages suitable for matrix manipulation. Even the most complex networks can be updated and resolved in a matter of minutes or even seconds on current PCs. For an analysis of the ICRB this was extremely useful because it meant that we could replicate our analyses across thousands of watersheds in the Basin limited only by our ability to develop the tables of input information characterizing watersheds and the management changes or trends expected with each alternative. That flexibility lends itself well to sensitivity analysis that allows an exploration of model behavior, changing alternatives, and competing worldviews.

Application to the ICRB

Our analyses focused on six salmonid fishes and their habitats. We considered effects of the management alternatives on habitat on all lands, but for salmonids we focused only on the potential spawning and rearing areas. We excluded consideration of areas classified only as corridors or seasonal habitats for three reasons: 1) spawning and rearing habitats are the critical areas found predominantly on Federal land and are the habitats most sensitive to Federal land-use management; 2) spawning and rearing areas are more likely to be in headwater systems; and 3) we have poor understanding and ability to predict the influence of multiple effects over the very large and complex catchments contributing to downstream habitats. We do not imply that federal land management does not influence more downstream areas, but that its effects will be more evident, predictable, and important in spawning and rearing areas.

For the purposes of our evaluation, we assumed that landscape conditions and management effects were immediate. Lotic habitats and fish populations are dynamic. They are continually responding to changes in landscapes as a result of management and natural processes. Some responses may occur quickly, whereas others may lag years or decades behind the changes on the landscape. Our results therefore cannot be viewed as absolute estimates of the number of strong populations or high capacity habitats that will exist. Rather, they are measures of the relative differences that may be expected among locations and species, given estimates of current and future landscape conditions associated with each alternative.

Network Structures

Within the limits of available information, we attempted to represent physical and biological processes most likely to influence distribution and dynamics of the salmonids and their habitats. Our networks were characterized by a collection of components (nodes) that represented environmental states or processes and two variables of primary interest, aquatic habitat capacity and the future status of each salmonid.

Within the geomorphic constraints of any watershed, upslope disturbances (e.g., logging, roads, fire) that cause accelerated production of sediment, alter hydrologic regimes, or alter the characteristics of riparian areas are generally accepted as primary drivers that influence the condition of habitats for fishes in wildland systems. In our network aquatic habitat capacity depended on: (1) generation and delivery of sediment; (2) the occurrence of large channel reorganizing floods; and (3) the condition or integrity of the riparian corridor. Sediment was defined as the relative amount of sediment entering streams above natural rates. It was influenced by *road density* and *ground disturbance* (i.e., logging, thinning and prescribed fire), topographic conditions (*slope steepness*), and management activities designed to mitigate erosion or sediment delivery (*standards and guides*). Hydrologic effects included probability of flood/debris-flow events that could reorganize large portions of the steam network. These events were influenced by the probability of having a flood-generating storm, by slope angle (*slope 2*) and the occurrence of large fires that dramatically alter hydrologic function (*fire-rain*). Riparian condition was perceived as those characteristics influencing shading and climate moderation, bank stabilization and water storage, and delivery of coarse and fine organic material. It was influenced by *prior riparian condition*, *future grazing*, and management activities intended to conserve or restore riparian function (*standards and guides*).

Salmonids can be strongly influenced by the physical capacity and quality of habitat. Status of many populations, however, is not strictly a function of local habitat conditions. Introductions of exotic species or the loss of some keystone forms such as the anadromous salmonids may have profound effects. Larger scale habitat fragmentation and isolation from surrounding populations can also be important.

For these reasons the future status of the salmonids in each subwatershed was conditional on one or two factors that were independent of local habitat capacity. The future status of resident salmonids (bull trout, Yellowstone and westslope cutthroat trout, redband trout) depended upon *aquatic habitat capacity* and on *biological potential* of the existing population. In addition to these two, status of anadromous salmonids (stream type chinook salmon, steelhead) also depended on conditions influencing survival of migrants to and from the ocean (*migrant survival*). Biological potential represented constraints on population resilience, productivity, and size, dependent on current condition of the population (*current population status)*, the biotic effects associated with introduction or loss of members of the associated fish community (*exotic threat, anadromous loss)* and potential for demographic support from surrounding populations (*refounding and support, connectivity)*. Migrant survival depended both on conditions in the migratory corridor (i.e., number of dams that must be passed by migrating fish) and in the ocean.

Conditional dependencies in the aquatic habitat and population status networks were estimated using a combination of expert judgement and empirical relationships. The conditional probabilities linking directly with salmonid status were estimated independently for each species. For many of the key nodes we relied on multiple experts and averaged their estimated probabilities to reflect the relative uncertainty in collective beliefs. Any disagreement among experts produced a more uniform distribution of probabilities across states reflecting greater uncertainty in the conditional dependencies.

Available Information

Existing syntheses of landscape characteristics, fish assemblages, and an interpretation of planned management activities based on the alternatives outlined in the Supplemental Draft Environmental Impact Statement (SDEIS) represented the primary information available for our analyses. The biophysical coverages summarized to subwatersheds were obtained from the Interior Columbia Basin Ecosystem Management Project. Predictions of the land management activities included estimates of road density, mechanical ground disturbance, livestock grazing, and probability of large wildfire. In addition, we developed our own series of rules to assign a level of conservation and restoration (high, moderate, low) in each subwatershed based on the management direction outlined in the SDEIS alternatives.

All inputs for the networks were summarized from equivalent (species status and distribution) or finer resolution (landscape data derived at 1 km pixel) information. Variables represented by the nodes in our network then are viewed as conditions representative of entire subwatersheds.

Plan for Analyzing and Testing Hypotheses (PATH)

Our primary focus in the work accomplished to date has been to make a start at developing some tools and procedures for conducting quantitative analyses of experimental actions. We have developed a set of experimental management (EM) modeling tools that allow us to quickly assess the biological/conservation consequences and learning opportunities of actions that affect overall survival of Snake River spring and summer chinook salmon. These models are intended to provide a starting point for additional work after PATH is discontinued.

We have used these models to conduct some *preliminary* screening and analyses of the short-list of actions. These analyses are preliminary because:

1. We have not done a thorough assessment of the feasibility of implementing these actions. Because of this, we have evaluated a set of generic and hypothetical experimental actions without speculating about how these actions might be actually implemented.
2. We have only looked at the effects of individual actions; combinations of actions may be more effective.
3. We assume that an action will have some effect, then assess the resulting biological and learning consequences. We have not assessed the weight of evidence in support or against the assumed magnitude of effects.
4. In most cases, we have only looked at how long it would take to detect effects in overall survival, from spawner-recruit data.

Our preliminary assessments should therefore be viewed as illustrations of “what if” scenarios of management experiments. We address the question “Suppose that a particular action could be feasibly implemented and had a particular effect on Snake R. spring/summer chinook populations: What would the biological consequences of such an action be, how difficult would it be to estimate that effect from spawner-recruit data with reasonable confidence, and what are the resulting trade-offs between learning and biological objectives?” These assessments are useful for developing and testing our EM models, and for providing some broad guidance on the learning and conservation implications of various actions.

Outputs

A. Biological

The primary output of the model is projected numbers of spawners and recruits for seven Snake River index stocks of spring / summer chinook. From these, we calculate probabilities of exceeding 1995 BiOp recovery and survival thresholds[1] over 24 and 100

1 These are the probabilities that the number of spawners of 6 out of the 7 index stocks will exceed survival and recovery threshold numbers of spawners. Survival thresholds range from 150 to 300 spawners; recovery thresholds range from 350 to 1150 spawners, depending on the stock.

years (survival standards) and 24 and 48 years (recovery standards). We also calculate the probability of going to one spawner or less in a given year (over 10 and 100 years) as a quasi-extinction metric similar to that used by CRI in their August 1999 document.

In order to calculate these metrics, we assume that actions will be maintained for the duration of each metric's time horizon (i.e. 24 and 100 years for survival probabilities, 24 and 48 years for recovery probabilities, and 10 and 100 years for quasi-extinction metrics). With the possible exception of the drawdown actions, this assumption is probably not realistic because if one discovers a suite of actions that meets survival and recovery requirements, one likely would not continue with the original on/off experiment. The population metrics included here may thus be viewed as a relative index of the biological consequences to the stocks, if the experimental actions were continued indefinitely.

Probabilities of exceeding survival and recovery thresholds are lower in this analysis than in previous PATH reports because of differences in some of the assumptions and data used in the model:

- Because we use 1978-1994 as representative of current conditions, we are assuming that the poor ocean conditions that existed in this time period continue into the future.
- We have assumed in most cases that extra mortality[2] is "here to stay". That is, we assume that the same high level of extra mortality that was experienced in 1978-1994 continues on into the future.
- This analysis uses updated spawner-recruit data, which includes spawner data up to 1999. Spawner numbers in these years were generally low, with zero spawners in some years for Marsh Creek and Sulphur index stocks.

B. Learning

The main metrics of how much can be learned from an action are expressed in terms of the probability of estimating effects of an action over various time frames, or, conversely, how long it would take to estimate an effect with a certain level of confidence. Various criteria can be applied to determine how long an experiment needs to be run to estimate effect sizes that reflect the risk preferences of decision-makers. We present three examples for illustration:

1) one approach might be to require the experiment to <u>not</u> have a <u>negative</u> estimated effect on survival. In this case, decision-makers would want to know the probability of estimating any non-zero effect on survival rates, and how this probability changes as the experiment goes on. This is the least stringent of the three examples; the effect can be estimated with high probability in a relatively short period of time.

---

2 Extra mortality is defined as any mortality occurring outside the juvenile migration corridor that is not accounted for by: (1) productivity parameters in the spawner-recruit relationship; (2) estimates of direct mortality within the migration corridor; (3) common year effects influencing both Snake River and Lower Columbia River stocks; and (4) random effects specific to each stock in each year.

2) decision-makers may want to know that the estimated effect of the action is close to (say, 80% of) its hypothesized effect. When hypothesized effects are large, this is generally the most difficult criterion to meet (i.e., probabilities of meeting it are lowest).

3) if one applies standard criteria for designing experiments, we would want to be fairly certain that we do not claim that an effect exists when in fact the action has no effect. To do this, we define a critical effect size (Δm^*), which is set at a level that minimizes the probability (0.05 or less) of incorrectly concluding that there is an effect when in fact there is none. The probability of detecting this critical effect size, if it exists, is called the "power" of the experiment; the higher this probability, the more "powerful" the experiment. This is the most difficult criterion to meet when hypothesized effects are small.

<u>Model Structure</u>

The model is based on the Ricker model of Recruits vs. Spawners that is used for most analyses of Pacific salmon populations. Natural log units are used to linearize this model because this makes it easier to deal with the wide variability that characterizes most spawner-recruit data sets and to estimate the model's parameters[3]. The model can be expressed as:

	$\ln(R_{i,t}/S_{i,t}) =$	a_i +	$b_i S_{i,t}$	+ m_t	+ $\varepsilon_{i,t}$
		⇓	⇓	⇓	⇓
or alternatively as:	R/S $\propto$	productivity factor	carrying cap. factor	year effect	error term
	for	each stock, all years	each stock, all years	all stocks, each year	each stock, each year

These parameters are estimated from historical spawner-recruit data, then used in forward projections to simulate the effects of actions. Assumptions about the effects of experimental actions are implemented in the model through the "m_t" or "year effect" term, which can be thought of a general survival factor for each year that affects all Snake River spring chinook stocks simultaneously. In the model, m_t values are calculated relative to the average survival rate from spawner to recruit over the entire historical time period (1958 to 1994). For years when $m_t = 0$, overall survival was equal to the long-term average. When m_t is positive, overall survival was better than average; when m_t is negative survival was worse than average. Because m_t is in natural log units, every unit increase (decrease) in m_t increases (decreases) survival by a factor of 2.7 (1 / 2.7) relative to the historical average. For example, when $m_t = 1$, survival in that year was 2.7 times the historical average. When $m_t = 2$, survival in that year was 7.4X the historical average (=2.7 X 2.7). When $m_t = -1$, survival in that year was 0.37X the historical average (=1 / 2.7).

3 When natural log units are used the error term, which for spawner-recruit data is assumed to follow a log-normal distribution, is transformed into a normally-distributed parameter. This allows us to fit a linear model to the log-transformed data.

Modeling Process

1. Estimate model parameters (a_i, b_i, m_t, $\varepsilon_{i,t}$) from historical spawner-recruit data (Figure ES-1).

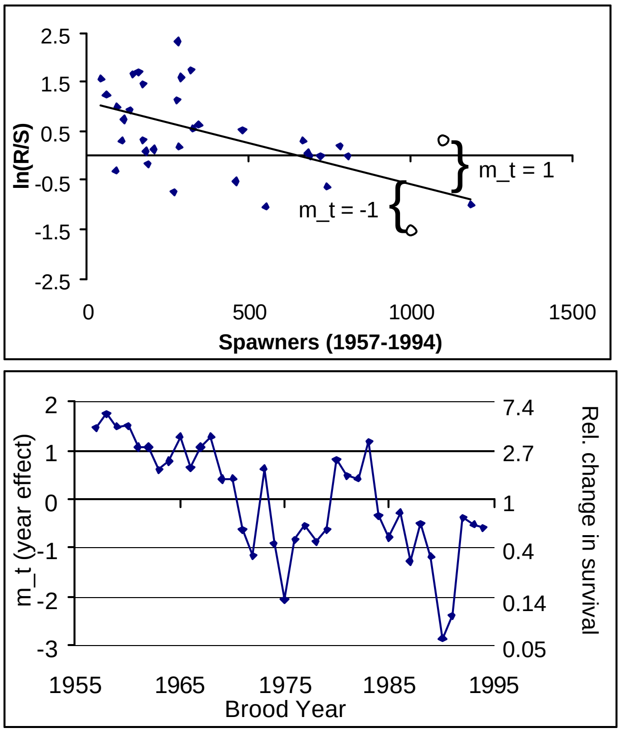


Figure ES-1: Estimation of m_t from historical data (left) and historical time series of m_t (right). Spawner-recruit data shown in left panel is a hypothetical dataset generated for illustration purposes.

2. Specify a future time series of m_t for simulating experimental actions. The future time series of m_t =

   an historical m_t value selected at random from the 1978-1994 m_t values (this was used as the base period because conditions between 1978 to 1994 were assumed to be most like present conditions)

   plus

   a hypothesized effect on survival of the future action (this term is called Δm). For example, consider a hypothetical experiment in which some action is turned on and off in successive years. If this experimental action is hypothesized to cause a 2.7-fold improvement in survival in each year the action is implemented ("treatment year") relative to years where the treatment is not applied ("control year"), the time series of Δm values for the forward simulation would be Δm = 1, 0, 1, 0, etc. for the duration of the experiment.

The result is a future time series of Δm values that shows how an action is hypothesized to change overall spawner-recruit survival rates from the survival rates experienced between 1978-1994 (Figure ES-2 shows an example using the Δm=1/0 in on/off years example).

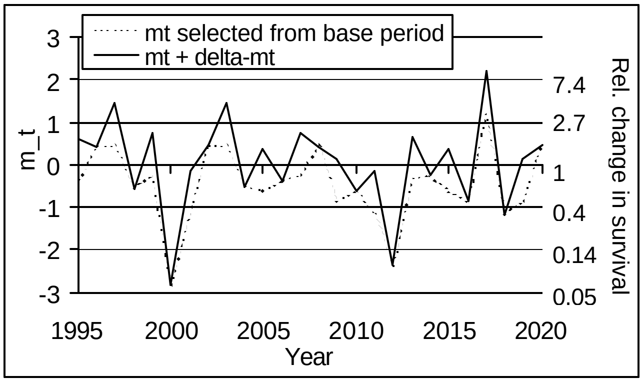


Figure ES-2: Example future time series of m_t values for forward projections of Δm=1,0 on/off experiment.

3. Use the future time series of m_t, along with historical estimates of the other model parameters (a, b, ε) to project populations through the experimental period. Simulate future data collection and analyses. Estimate Δm as the difference in average simulated ln(R/S) in treatment and control years (Figure ES-3). Calculate probabilities of recovery, survival, and extinction.

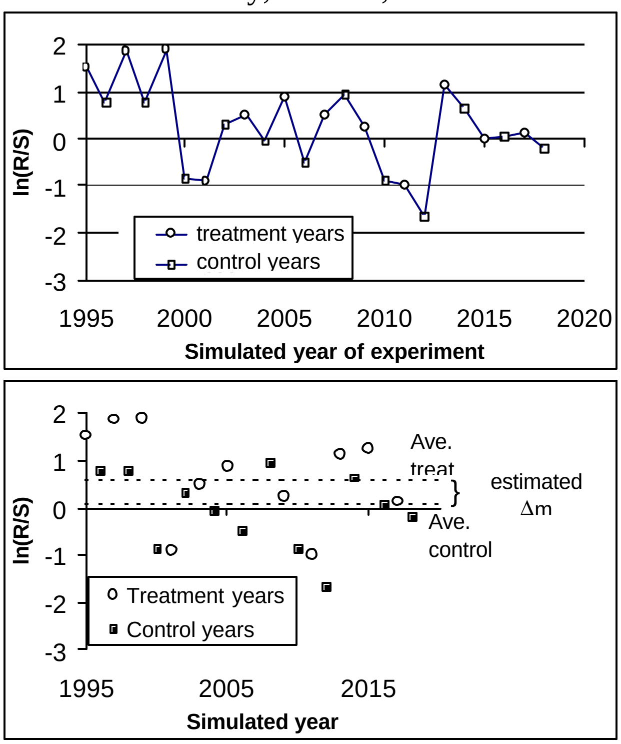


Figure ES-3: Method of estimating Δm from future time series of ln(R/S) data. Data are hypothetical examples for illustration purposes only.

4. Do this over multiple trials (i.e., many alternative futures) to get a frequency distribution of estimated Δm and biological metrics for different lengths of experiments (longer experiments = more data = better information) (Figure ES-4). Calculate probabilities of detecting various levels of Δm. Earlier in Section ES.3 we presented three examples of effects decision-makers may be interested in. These effect sizes can be translated into terms of Δm (Table ES-2). The frequency distributions are used to calculate the probabilities of estimating these Δm values.

Table ES-2: Δm equivalents of three example effect sizes decision-makers may be interested in estimating.

Effect	Corresponding Estimated Δm value
Experiment has no negative effect on survival	$\Delta m \geq 0$
Effect of the action is close to its hypothesized effect	$\Delta m \geq 0.8$ X the "true" hypothesized Δm value
Statistical "critical" effect size (Δm*)	$\Delta m^* \geq 1.64$ X std. deviation of the estimated Δm

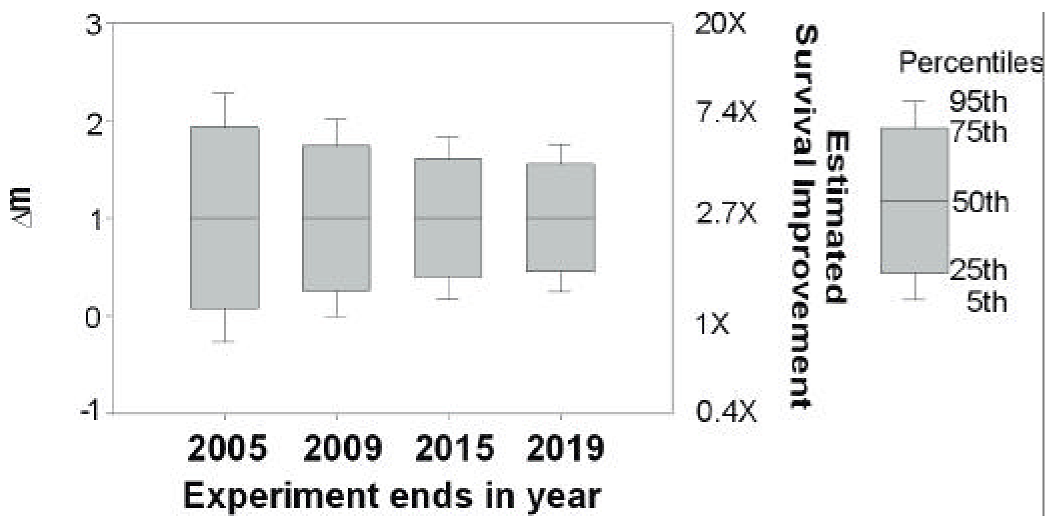


Figure ES-4: Distribution of estimated Δm values for a 1/0 on/off type of experiment of various durations.

<u>Inputs</u>

The primary model inputs are time series of Δm values for each experimental action, where these Δm's represent hypotheses about how the action will affect overall survival rates relative to those experienced from 1978 to 1994. These are specified for a series of "generic" actions, in addition to the six experimental actions.

Appendix III. Written Responses of the Principal Scientists to the ISAB's Questions about Their Analytical Approaches

Contents:

In undertaking this project, the ISAB requested the assistance of the principal scientists involved in each analytical approach. The ISAB asked that they: 1) provide written responses to a series of questions about their modeling effort, 2) give an oral presentation of their responses and participate in a discussion of the responses with the ISAB, and 3) participate in a panel discussion with the ISAB and the representatives of each of the modeling efforts. The principal scientists were informed that their responses would be incorporated into and appended *verbatim* to the ISAB report. Each of the scientists provided written responses, which were distributed to the other participants in the May 2000 workshop at the Northwest Power Planning Council offices in Portland. This appendix contains the responses.

The ISAB asked the principal scientists to address the following questions in their written responses and oral presentation:

1. What is the purpose of your modeling effort? What questions or problems were your models designed to address?

2. Summarize the major conclusions of your modeling effort relative to the four H's: habitat, harvest, hydropower, and hatcheries. Specifically, state your model's conclusions relative to the following:

a. The efficacy of dam breaching or drawdown to natural river levels for delisting ESA species and restoring diverse and productive populations of native fishes throughout the Columbia River Basin?

b. The efficacy of hatcheries for delisting ESA species and restoring diverse and productive populations of native fishes throughout the Columbia River Basin? Consider hatcheries that mitigate for lost habitat and those used to supplement depleted stocks.

c. Allocation of harvest and harvest levels needed to delist ESA species and restore diverse and productive populations of native fishes throughout the Columbia River Basin?

d. The efficacy of restoration of tributary and mainstem habitat for delisting ESA species and restoring diverse and productive populations of native fishes throughout the Columbia River Basin?

3. What kinds of information or data are needed to run your model?

4. What are the strengths and weaknesses of your model?

5. What are the assumptions of your model?

6. How does your model address uncertainty?

7. All models make predictions. Why do you think your model's predictions are accurate?

8. How does your modeling effort relate to or contrast with the other three modeling efforts?

9. What advice would you give decision-makers on how they should use your model to support decisions regarding salmon recovery in the Columbia River Basin?

ICBEMP Answers

Submitted by:

Bruce Rieman, USDA Forest Service, Rocky Mountain Research Station

Philip Howell, USDA Forest Service, Pacific Northwest Research Station

James Peterson, Cooperative Fish and Wildlife Research Unit, University of Georgia

1. What is the purpose of your modeling effort? What questions or problems were your models designed to address?

Our models were developed to evaluate the relative effects of proposed land management plans on aquatic habitats and species on Forest Service and Bureau of Land Management lands in the Interior Columbia Basin. Specifically, our models were used to compare the expected changes (from current conditions) in aquatic habitat capacity and future population status of six salmonid species among three land management alternatives. The species examined are relatively widely distributed on FS/BLM and included bull trout, westslope cutthroat trout, Yellowstone cutthroat trout, redband (resident rainbow trout), stream-type chinook, and steelhead. Our results were projected at 10 and 100 years from present and summarized for 6th code hydrologic units containing FS/BLM lands, all lands (public and private), and other subdivisions (e.g., ESUs) within the ICBEMP management area. Relative trends for projected habitat capacity of the 6th code watersheds associated with 17 other native fishes classified as sensitive (Lee et al. 1997) were also summarized, but we did not project population status for these species. The Supplemental Draft Environmental Impact Statement describing the model in detail will be released for public comment in the near future.

2. Summarize the major conclusions of your modeling effort relative to the four H's: habitat, harvest, hydropower, and hatcheries. Specifically, state your model's conclusions relative to the following:

a. The efficacy of dam breaching or drawdown to natural river levels for delisting ESA species and restoring diverse and productive populations of native fishes throughout the Columbia River Basin?

Our models are not useful for evaluating the efficacy of dam breaching or drawdown. Dam effects were included in our models because our direction was to provide as much spatial detail as possible and because dams could potentially obscure the effects of changes in spawning and juvenile rearing habitat associated with each of the alternatives. Our models reflected a strong belief that mainstem dams have an important influence on migrant survival that may constrain the response of populations to habitat conditions.

We evaluated two scenarios: one in which migrant survival in Snake Basin chinook and steelhead areas was classified as low and another where migrant survival was classified as moderate. Even though the manipulation of migrant survival produced positive changes in some outcomes in the status of chinook and steelhead, the relative performance of the SDEIS alternatives did not change. Although our models reflected the belief that dams have an important influence on survival, that influence had essentially no effect on the projected differences among the alternatives we were asked to evaluate.

b. The efficacy of hatcheries for delisting ESA species and restoring diverse and productive populations of native fishes throughout the Columbia River Basin? Consider hatcheries that mitigate for lost habitat and those used to supplement depleted stocks.

The efficacy of hatcheries was not addressed in our models.

c. Allocation of harvest and harvest levels needed to delist ESA species and restore diverse and productive populations of native fishes throughout the Columbia River Basin?

The allocation of harvest levels was not addressed in our models.

d. The efficacy of restoration of tributary and mainstem habitat for delisting ESA species and restoring diverse and productive populations of native fishes throughout the Columbia River Basin?

Our models are useful for considering spatially explicit trends that may result through habitat and land use management on Federal lands. We do not believe it is possible to predict the absolute change in habitat capacity or quality, or the absolute change in abundance or productivity of the fish stocks associated with these lands at this scale of analysis.

Our models focused on the effects of Forest Service/BLM land management and restoration of spawning and initial rearing habitat for six salmonids fishes in the Columbia Basin east of the Cascade Range and outside of federal lands managed under the Northwest Forest Plan. Although we considered all public and private lands within the ICBEMP management area to provide a larger context, the alternatives and their outcomes varied only on FS/BLM lands. The relative effects of proposed federal habitat conservation and restoration on listed and unlisted species were also influenced by the species' biological potential (e.g., current status, threat of exotics, refounding and support of adjacent populations) and mainstem migrant survival for anadromous species.

Although the outcomes varied by alternative, species, and strata, all three SDEIS alternatives projected positive long-term trends relative to current conditions in aquatic habitat and to a lesser degree, in species status. All of the alternatives were projected to maintain core areas, where populations are currently strong and more widely distributed. We projected some rebuilding of depressed populations, especially in watersheds identified for restoration emphasis. Although habitat conditions were expected to improve under each alternative, there was more uncertainty associated with the responses of many populations due to constraints of non-habitat factors (e.g. isolation, exotic species, migrant survival). Some declines in habitat conditions were projected in the short term under alternative 3.

3. What kinds of information or data are needed to run your model?

Predictions or coverages of the following for each 6th code HUC on Federal Lands within the Interior Columbia River Basin analysis area:

- Road density
- Slope class
- Proportion of the area influenced directly by ground disturbing activities (e.g. logging, skidding, prescribed fire) per decade
- An interpretation of the level of habitat mitigation and restoration described or implied in the management alternatives
- Grazing intensity
- Past management history (e.g. wilderness, intensively or lightly managed forest)
- Probability of large wildfire
- Occurrence of introduced fishes
- Status of existing populations of the particular species modeled
- Where current species status is ‘absent’, a prediction of potential for spawning and rearing
- Status of populations in surrounding 6th code HUCs
- Connectivity among 6th codes HUCS within the encompassing 4th code HUC
- Number of dams that must be passed by anadromous fish

4. ***What are the strengths and weaknesses of your model?***

Past attempts to evaluate the alternatives of the previous ICBEMP DEIS and similar large scale management proposals, such as FEMAT, focused on arraying available information (e.g., management direction, projected levels of activity, knowledge of fish or habitat distribution and status) and then asking one or more experts to formulate opinions about the likely future status or trend of habitat and populations across the affected range. This was an unsatisfying and contentious approach for several reasons. The interpretations represented a synthesis that was subjective and not logically explicit. It was difficult to conceptualize and account for the complexity of the management direction and the multiple interacting physical and biological effects. Although assumptions could be stated, it was hard to determine how those assumptions may have influenced the outcome or uncertainty in the outcome. Because the analysis was based on individual interpretations of all of the data, it was difficult to replicate or update the analysis when the management alternative or key assumptions were modified. The results were not easily quantified and were not spatially explicit. Management and its effects were generalized across a large heterogeneous landscape. It was difficult to evaluate the relative differences among alternatives and the uncertainty in any predicted outcome.

We believe our approach provided several distinct advantages. Complex physical and biological interactions and management alternatives could be compartmentalized into simpler, more comprehensible components. By formalizing our understanding and assumptions, we provided a framework for exploration of differences in the management alternatives that is quantifiable, spatially explicit, and flexible. The networks can incorporate both quantitative and qualitative information, which is essential to any analysis attempted at this scale. Our assumptions and model relationships can be tested, and the analysis repeated in a consistent, expedient manner. Critical uncertainties for management can be determined, and the structure of the networks can be changed to explore the sensitivity of results to those uncertainties.

Despite these benefits these networks have important limitations. Our analysis is limited by the subjective nature of the information available to us, by the quality or accuracy of the data that do exist, and by our understanding of and ability to describe key linkages in aquatic ecosystems at this scale. In many cases, the landscape data represent only surrogates or correlates of key elements or processes. Often the interpretation of any process was based on models requiring extrapolation across spatial and temporal scales. We view the networks as very simple abstractions that represent our attempt to capture the way we think about systems and the relative importance of the different processes influencing fish. They are a relatively simple means of replicating a professional interpretation of conditions and effects in a consistent way across a very complex landscape. They may exaggerate some patterns and under represent others. Uncertainty is explicit in the use of conditional probabilities. This uncertainty reflects the limitations of understanding and information but also means that trends and differences can be obscured. Because much of the information represented in the networks is subjective and represents the strength of the belief in those relationships, the results can be viewed only as relative trends among alternatives and not as predictions of the absolute number or true probabilities of high quality watersheds or extant populations.

5. What are the assumptions of your model?

Our key general assumptions are listed below. Other assumptions related to interpretation of specific elements of the alternatives or the model are described in other analysis documents. There are many assumptions inherent in the development of the model structure and the parameter estimates (i.e., conditional probability tables [cpts]) that are not captured in this list. In some cases, particularly where we used expert judgment to develop the cpts, those assumptions are captured in the model but are not described in the model documentation.

We assumed that the interpretation of the SDEIS and resulting predictions of landscape characteristics and disturbance provided by the landscape team is accurate in value and spatial representation. We similarly assume that information derived from the ICBEMP Scientific Assessment (e.g., Quigley and Arbelbide 1997) and available in existing basin coverages is accurate. We know that errors exist in the data and in some cases the relative magnitude of the error is not quantified. However, we assumed that those errors do not meaningfully compromise the results of the analysis.

We assumed that decreases in road density reflect the actual removal of roads and most of their related adverse effects on the landscape. It is recognized that removal may not include re-contouring if inappropriate, but does include re-vegetating and no vehicular use and the restoration of hydrologic function.

Our definition of aquatic habitat capacity implies that a range in habitat conditions is possible even without human disturbance and that habitat will vary through time in response to natural disturbance and vegetation succession. We do not assume that optimum conditions always will exist in the absence of human activity. We did assume that a subwatershed (HUC 6) in which sediment input, riparian habitat, and hydrologic regime have not been substantially altered by human activity will be more likely to contain aquatic habitat conditions that are closer to optimum for indigenous salmonid species than a subwatershed where one or more of these components have been considerably altered by human activity.

We assumed that aquatic restoration needs are apparent and that the habitat restoration methods to be used will be effective.

We assumed that the each of the alternatives would be fully implemented to achieve the objectives and meet the standards as described.

We assumed that field units will be staffed with adequate aquatic expertise to effectively implement analysis, conservation, and restoration direction.

We assumed that there is greater uncertainty regarding the effectiveness of restoration compared to conserving existing high quality aquatic habitat.

Some point source impacts, such as mining, could adversely affect aquatic habitat condition under all alternatives, but there were inadequate data to determine their potential effects in a spatial context.

We assumed that interim FS direction for roadless areas will be replaced by comparable long-term direction for FS lands. That direction primarily restricts road building, but does not change the management allocation. For example, timber harvest and other uses could still occur but would be limited to access not dependent on new roads. It was assumed the existing minimal level of road construction on BLM lands will continue.

Even though the belief networks employ probabilities, we do not assume that they are accurate estimates of true probabilities or "risks." Rather, they represent the strength of our belief in the status or trends for particular elements of the system.

We assumed that the differences among experts that estimated conditional probabilities in our networks represented uncertainty in outcomes resulting from inherent variability in the system and uncertainty in our understanding of nature. Differences or confusion in the interpretation of or the definition of states of nature reflected the limitations of our understanding.

It is known that the predicted results of management activities and alternatives at 10 and 100 years will continue to be influenced by succession and evolution of the system into the future. For the sake of the evaluation, however, we assumed that the biological response was an immediate result of the landscape conditions at the point of evaluation (i.e., 0, 10, 100 years).

Where 6^{th} code HUCS (subwatersheds) were not true watersheds but composites of a high order mainstem reach and low order tributaries we assumed that habitat conditions for fish in the subwatershed being evaluated were represented by the mean of conditions in contributing subwatersheds.

We assumed that the effects of federal land management activities on salmonid fishes will be most influential and measurable in spawning and rearing habitats. Our analysis was limited to subwatersheds classified as existing or potential spawning and rearing habitat. We assumed then that Federal land management does not significantly influence the condition of populations outside these areas.

We assumed that the status and trends of salmonids and their habitats are suitable indicators of the conditions in aquatic ecosystems in general related to federal land management.

We assumed that climate change would not influence the relative differences among the alternatives.

We assumed that factors influencing the condition of habitats for fishes that are not explicitly represented in the models are incorporated as additional uncertainty about the likely future status.

For subwatersheds with multiple ownership or management direction, we assumed that the net effect of the mix of prescriptions across the different management areas was the area weighted average of the probabilities associated with each.

We assumed that the uncertainty associated with each estimate of subjective probability in the cpts is trivial compared to the overall uncertainty in the model, and hence treat each of these probabilities as point estimates. Further, we did not weight the information provided by individual experts and, therefore, assumed that each had the same level of relevant knowledge.

We assumed that planning and analysis at levels of organization below the Interior Columbia Basin (e.g. subbasin review and watershed analysis or EAWS) would be necessary to effectively implement the direction in the SDEIS. We assumed that these analyses where required would be effective and resulted in the proper citing of conservation and restoration measures. Where these analyses were not required we assumed greater uncertainty, and less success in producing the intended outcome.

6. How does your model address uncertainty?

The uncertainty surrounding future outcomes was explicitly incorporated in our model through the use of probability. We modeled the relationships among states using conditional probabilities in the form of subjective (belief) and empirical probability estimates. All model inputs were probabilistic with the exception of average subwatershed slope, which was deterministic (i.e., known). The model outputs were expressed as the probability that future habitat capacity and salmonid population status were in each of 3 states.

7. All models make predictions. Why do you think your model's predictions are accurate?

We don't think our predictions are accurate in the absolute sense. That is we don't believe the probabilities we estimate for a particular state in habitat condition or species status reflect true probabilities. As stated in #4, our models relied, to a great extent, on the subjective belief of experts due to the complex nature and lack of empirical data and models for estimating the effects of land management at large scales and over long time frames. Thus, the models should be considered as explicit formalized thought-processes rather than empirically based predictive models. Because we sampled a wide array of experts in the physical and biological sciences, we believe that the models are "accurate" in that they faithfully reflect the collective beliefs of experts as to the influence of land management on aquatic habitats and salmonids. Human judgments of subjective probability, however, are often inconsistent with probability laws and definitions. In contrast, the empirically derived components of our model were relatively accurate, with an average cross-validation error rate of 21%, across species and components.

8. How does your modeling effort relate to or contrast with the other three modeling efforts?

It appears to us that each effort had very different objectives.

Our models were designed to analyze the relative effects of the SDEIS alternatives on freshwater salmonid habitat capacity, particularly on federal lands, and future population status of both resident and anadromous salmonids influenced by FS/BLM management. They were not designed to assess the risks of all of the factors affecting salmonids in the Columbia basin throughout their life cycles and various life histories or the risks associated with the Snake River dams. PATH and EDT are large, complex models, whereas the effects of mainstem dams in our models are reduced to a single variable and associated set of probabilities generated by an expert panel. In essence that component of the system that other groups have tried to analyze through very complicated approaches was distilled to some very broad assumptions in our analysis. Our panel assumed that dams have an important influence on the survival of salmon and steelhead that can in some cases outweigh the effects of habitat--our models did not tell us that. Based on some of the CRI results, the assumption used in our model may have been too conservative. Based on other analyses they may have been appropriate. Either may be right. It was not our intent to enter that debate or to attempt to resolve the uncertainty. Because of the uncertainty, we conducted two sets of analyses with a range of assumptions regarding the influence of the Snake River dams. Regardless of the assumptions used, the trends among the alternatives we were asked to evaluate remain the same. Although differences in those assumptions make a large difference in the absolute number of salmonid populations predicted to be present through our models, they have no effect on the relative differences among the alternatives. We emphasize that the absolute numbers are not particularly meaningful in interpreting our results. Furthermore, any differences or similarities among our models and other salmonid models for the Columbia in terms of how migrant survival is modeled have no effect on the projected outcomes of the alternatives on freshwater habitat capacity of anadromous species or other fishes on FS/BLM lands, where the alternatives are most directly influential. The original aquatic science assessment (Lee et al. 1997), subsequent published papers, and the SDEIS evaluation have all stressed the importance of maintaining and restoring freshwater habitat to the persistence of anadromous species, regardless of the relative impact of migrant survival or other Hs.

9. What advice would you give decision-makers on how they should use your model to support decisions regarding salmon recovery in the Columbia River Basin?

As stated in #1, our models were designed to evaluate the relative merit of 3 proposed management alternatives for FS/BLM lands in the Basin by examining their (relative) potential effects on the habitat of salmon and other fishes on those lands. That is, our models were specifically created to help decision-makers select a management plan for FS/BLM lands. The consequences of those activities on salmon recovery, although an important consideration, will depend on many factors outside of the scope of our models and the decisions made concerning management of FS/BLM lands. Our models should be used in concert with other models and information to weigh the merits of possible decisions affecting management of other public and private habitat, hydropower and hatchery operations, and harvest.

References Cited

Lee, Danny C.; Sedell, James R.; Rieman, Bruce E. [and others]. 1997. Chapter 4. Broadscale assessment of aquatic species and habitats. In: Quigley, Thomas M.; Arbelbide, Sylvia J., tech eds. 1997. An assessment of ecosystem components in the interior Columbia basin and portions of the Klamath and Great Basins: volume 3. Gen. Tech. Rep. PNW-GTR-405. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 4 vol: 1057-1713. (Quigley, Thomas M.; tech. ed.; The Interior Columbia Basin Ecosystem Management Project: Scientific Assessment).

Quigley, Thomas M.; Arbelbide, Sylvia J., tech. eds. 1997. An assessment of ecosystem components in the interior Columbia basin and portions of the Klamath and Great Basins. Gen. Tech. Rep. PNW-GTR-405. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 4 vol (Quigley, Thomas M., tech. ed.; The Interior Columbia Basin Ecosystem Management Project: Scientific Assessment).

Cumulative Risk Initiative Answers

Submitted by Peter Kareiva and Chris Jordan of CRI
NWFSC, NMFS

PROLOGUE

At the risk of repeating ourselves again, we would like to emphasize that the CRI is not just a modeling effort. While we realize that all science and all analyses are models, the CRI is best described as an organized chain of logic and analyses that for some steps relies on models. One distinguishing feature of the CRI is that it stops short of using numbers or calculations to make certain "leaps"; the CRI is unwilling to make quantitative leaps across data gaps that are too wide and too deep.

In a previous review of the CRI, the ISAB argued for moving towards a formal decision theoretic approach, and at the time of that review CRI researchers would probably have agreed with the ISAB. However, the CRI research team would now disagree for five intertwined reasons:

1. As we have become increasingly familiar with the data available for Columbia River salmon populations, we have concluded that the data are generally so poor that many of the pertinent management questions simply cannot be dealt with quantitatively (see the Executive Summary of McClure et al 2000).4

2. The core of the existing decision theoretic approach regarding Snake River dam breaching is an upstream/downstream comparison. However, there is clear statistical evidence that the different ESUs (upriver versus downriver chinook) respond non-uniformly to the same ocean variables (Levin, submitted), implying differences in time series cannot be attributed solely to dams even after a common environmental factor is accounted for.

3. Upcoming management decisions must concern the entire Columbia River Basin and its twelve listed ESUs. Yet a decision framework has only been attempted for the Snake River chinook salmon. It is impossible within even a one-year time period to imagine constructing a decision theoretic approach for the entire Columbia River Basin. Far more pressing is the need to synthesize and standardize data across ESUs and index stocks, and develop comparative approaches. Fundamental questions such as how many hatchery fish are spawning in the wild and what is their reproductive efficacy have far greater urgency than a decision analysis framework. In fact, without first answering such questions a decision analysis framework is likely to be unrewarding.

---

4 Included in our responses to the ISAB's Nive Questions are references to particular sections of a recent CRI analysis of eleven salmonid ESUs in the Columbia River Basin. The document, *A Standardized Quantitative Analysis of the Risks Faced by Salmonids in the Columbia River Basin* (referred to as McClure et al. 2000) is too lengthy to be read in its entirety as part of the ISAB project, but is included as part of our response to illustrate the scale and scope of the CRI effort. In addition, one can ground-truth our answers provided here by examining the relevant pages of this most recent CRI product. This document is available from the CRI website.

4. The emphasis on decision-theoretic approaches diminishes the emphasis on data. This is not a logical necessity, but it is appears to be a "tendency". Because modeling often gives the illusion of an "answer", there is reduced pressure to acquire or update data. For example, until recently run-reconstruction data for Snake River Chinook beyond brood year 1990 had not been updated, even though the data were available to update through brood year 1994. Ironically these most basic of all population data were four years behind the times while the region debated population-modeling strategy. The actual population numbers from recent years had become less important in the scientific discussion than modeling predictions about future population behaviors.

5. In its best manifestations, decision theory can provide guidance to policy-makers. However, policy-makers may not be able to evaluate the "firmness" or "credibility" of these answers. The CRI feels that the numbers from decision theory would represent an illusion of science when there is really very little science to support particular choices. Given the data, the CRI concludes that only bold management experiments can give scientific answers to the key questions. Of course this conclusion itself is the outcome of a decision process, but it is a matter of priorities and pragmatism as opposed to academic acumen. Given the many analyses that need to be done, a decision theoretic calculation to provide a "number" that says "*we do not know enough to know what to do, so do bold management experiments*", seems like a poor investment of scientific effort. Instead, the CRI will be emphasizing the design of monitoring programs, and exploring ways to estimate observation error because sampling error has been so neglected in the whole debate.

In summary, asking for a formal decision theoretic analysis at this stage is like asking a baby to walk before it can crawl. Such an analysis is a good idea, but there are many more pressing needs to attend to before any such analysis is attempted. PATH made a valiant effort at a decision theoretic component, and did a great service in synthesizing data, and exploring quantitative hypotheses about fish passage. But PATH floundered when it came to evaluating the merits of dam breaching relative to other management options. One main reason PATH has faced such a challenging task is that without being able to attribute upstream/downstream differences to dams, there simply may not be adequate data to contrast dam breaching to other management options within the same analytical framework.

In addition to the absence of the most basic data, a second problem with the Columbia River Basin salmon science arena is that while we can find numerous review chapters and perspectives papers, original analyses of actual data are remarkably scarce. Where are the published analyses linking habitat attributes to recruits per spawners? Where are the published analyses describing population trajectories in any quantitative manner? Where are the published analyses quantitatively examining hatchery impacts on wild stocks, as best as can be done? We suspect that the ISAB's recommendation of establishing a "decision theoretic framework" represents the same frustration we feel about an absence of analyses. Where we perhaps differ from the ISAB is in believing that we need to do a lot more basic data analyses before moving forward to a decision theory approach. To take advantage of ISAB's expertise, we have appended an EXCEL data file of time series of spawner counts. We encourage any or all ISAB members to try out their ideas by analyzing these time series using likelihood ratio or information theoretic methods, etc. Note that for many of these time series, we have no information on hatchery fish, or only crude guesses. In none of the time series has there been a direct estimate of the reproductive fitness of hatchery fish that spawn naturally.

Perhaps the shortage of clear data analyses exists because many of the analyses got bundled into large modeling efforts. Unfortunately, most of these analyses remain unpublished, poorly documented, and in our opinion impenetrable. The CRI is trying to initiate a culture of publishing in refereed journals, and publishing data papers, not just perspective or broad overview papers. Four noteworthy papers have been submitted in the last few months from the NWFSC efforts (not all are CRI papers, but they are related). We encourage the ISAB to contact the authors of those papers (their e-mail addresses are given below), and ask permission to review the work. We believe these papers are beginning to add some clarity that was previously lacking:

John.G.Williams@noaa.gov: updated downstream survival estimates and SARs for Snake River Chinook and Steelhead (using PIT tag data). Submitted to CJFAS.

Rich.Zabel@noaa.gov: a discussion of the temporal pattern of SARs in the Snake River and comparison of stocks in which the hydropower hypothesis of "extra mortality" is challenged. In press in CJFAS

Phil.Levin@noaa.gov: a critique of the upstream/downstream comparison, plus some statistical analyses indicating that different Columbia River Basin ESUs (including upriver versus downriver ESUs) respond differently to common ocean environmental variables

Jim Regetz, jregetz@princeton.edu: analysis of spatial variation in recruits per spawner for 22 chinook index stocks and the percentage of that variation explained by simple habitat variables. Submitted to Conservation Biology.

Michelle Marvier, mmarvier@scu.edu: numerical experiments with stage-structured matrices applied to Snake River Chinook salmon as a means of examining management options. Soon to be submitted to Science.

Phil.Levin@noaa.gov: an examination of competition between hatchery and wild fish as measured by ocean survival. Manuscript proceeding through the three levels of internal NWFSC review.

If more analyses had been submitted to journals three or four years ago, the documentation of the relevant science would be an order of magnitude better in the region – as it is, the lack of clear and well-written documentation is a major limiting factor in all discussions.

ANSWERS TO THE NINE ISAB QUESTIONS

1. What is the purpose of your modeling effort? What questions or problems were your models designed to address?

The CRI analysis is intended to synthesize data regarding the trends, dynamics and risks of threatened and endangered salmonids on the West Coast of North America. This scope (25 ESUs, ranging from California northward to the Puget Sound and the Canadian border) is considerably broader than even the Columbia River Basin. At this scale, the CRI will provide a standardized, yet flexible, framework within which recovery teams on the west coast can:

a. consider the status of their ESUs of concern;

b. examine the likely effects of recovery actions;

c. help design monitoring and evaluation strategies for determining how well actions are working and whether the ESUs as a whole are moving towards recovery.

2. Summarize the major conclusions of your modeling effort relative to the four H's: habitat, harvest, hydropower, and hatcheries.

The CRI analyses suggest that data and data analyses to address the merits of actions in the 4-H's are generally lacking, with the exception of harvest (in which case the analysis is straightforward since a harvested fish is unambiguously a dead fish). Nonetheless, the CRI does offer some analyses that are germane to a discussion of these four-H's.

Harvest: At the ESU level, it is possible to calculate the percentage increase in lambda[5] expected to result from particular reductions in harvest, using harvest rates from the time period during which lambda itself was estimated (up until brood year 1994). Performing this calculation makes it clear that by reducing harvest, the lambdas of three ESUs (Fall Chinook in the Snake River, Upper Willamette Chinook, Lower Columbia Chinook) could be sufficiently increased to well above 1.0, and the populations would be growing. Conversely, it is also evident that harvest reductions by themselves could not remedy the low lambdas in the remaining ESUs.

Hatcheries: Because basic data needed to assess the impacts of hatchery fish, including reproductive output of wild spawning hatchery fish, are lacking, it is impossible to obtain a satisfying answer to many questions regarding the impact of hatchery fish. Outside the Snake River Basin, we often lack even estimates for the relative frequency of hatchery fish on spawning grounds. As discussed in McClure et al. (2000), there are some data supporting a hypothesis that hatchery fish compete with wild fish in the ocean and under poor conditions can reduce the survival of wild fish (Levin, in review). If these analyses

---

5 The CRI measures population performance with "lambda", the annual rate of population change estimated in a manner that includes environmental variability, and as much as possible, accounts for sampling error. A lambda greater than one results in an increasing population, while lambda < 1.0 implies a shrinking population.

are on target, then it is possible that substantial improvements may be gained by alterations of hatchery policy. However, for many ESU's, due to the presence of hatchery fish in counts of natural spawners, it is difficult to assess the most fundamental demographic processes of populations.

Habitat: The CRI firmly believes that habitat restoration offers substantial opportunities for population recovery. A preliminary analysis of spatial variation in recruits per spawner data at the sub-watershed level revealed that a few simple habitat variables explained over 60% of the variation among chinook stocks in their productivity (Regetz, in review). The same 22 stocks did not reveal any such relationship with respect to simple hydropower variables such as the number of dams between the spawning ground and the ocean (which varied from 1 to 9). This is not evidence that dams and hydropower are unimportant, but rather puts the success of the habitat regression model in the context of other explanatory variables. A second analysis, better connected to specific parcels of land within ESUs, has been provided by the Watershed Processes Program of the NWFSC (section V.F.2 of McClure et al 2000). In this analysis salmon productivity was found to be related to good quality riparian zones in non-forested areas, low water temperatures, and low gradients.

Hydropower: Hydropower operations are usually discussed with respect to Snake River Spring/Summer Chinook salmon, although, given the widespread presence of dams throughout the Columbia Basin, they clearly impact all of the ESUs (see section V. of McClure et al. 2000). Clearly the hydropower system can alter mortality. However, rather than searching for extra mortality that must be attributed to some external cause or disease, it would be more parsimonious to ask the following questions:

By how much is dam breaching likely to increase downstream survival, upstream survival and estuarine/ocean survival? What is the evidence for such increases and what are their consequences in terms of improved lambdas?

There are good data to show that downstream survival may actually decrease following dam breaching because the in-river survival already comes close to matching pre-dam conditions (Williams et al 2000), and because transport survival is very high. Upstream survival is likely to increase by 15-30% with breaching, depending on estimates and interpretation of current conversion rates. The big unknown, of course, is the expected increase in ocean/estuarine survival due to removal of the four dams. Since the CRI concludes this is such an important uncertainty, it ran analyses of potential increases in survival that ranged from 20% to 180% in increments of 20% (see figure VI-5 of McClure et al.). It appears that dam breaching would need to increase early ocean/estuarine survival by 100% or more in order to increase lambda enough that extinction risk were mitigated. It is worth noting that an increase of 100% corresponds to fish that are one half as fit because of their travel through dams – a magnitude of depressed fitness that ought to be detectable on an individual fish basis. As a result of these analyses the CRI suggests that a future emphasis on fitness measures of individual fish (especially in conjunction with PIT tag data) may provide a more tractable indicator of whether dam breaching could be expected to recover spring/summer chinook salmon in the Snake River.

3. What kinds of information or data are needed to run your model?

All CRI analyses start with time series of spawner counts to estimate recruits per spawners or annual rates of population growth. In the few data rich cases the CRI approach is to develop demographic projection matrices. Some estimate of the age structure of returning fish is needed, but the extinction modeling and estimation of lambda are robust to large and erratic errors in this age estimation. Hence the most critical data are spawner counts. It is worth noting that even this most basic information is often lacking, or is compromised by the presence of hatchery fish (see section V. of McClure et al. 2000).

A second piece of critical information the CRI would like to have, or be able to estimate, is observation error. The CRI will make a concerted effort over the next several months to provide bounds to the magnitude of observation errors in spawner counts and will then explore the implications of this uncertainty for monitoring programs (see discussion of "detectability in section V.C of McClure et al, 2000).

4. What are the strengths and weaknesses of your model?

Weaknesses:

a.) The CRI does not provide answers in the form that policy makers seem to desire. Policy makers want simple numbers – how many fish will we get if we do X? It is the CRI's contention that given the quality of existing data any such numbers would be largely be "guesses" pulled out of thin air.

b.) The CRI is not a formal decision theoretic approach.

c.) Since the CRI focuses on data from 1980 onward, it does not cover an entire Pacific Decadal Oscillation and hence does not incorporate ocean cycles or trends in ocean conditions. Because recent years have reflected the poorest ocean conditions of the modern record, this weakness means that the CRI is likely to overestimate risks if one assumes that ocean conditions will soon improve. However, we feel that this over-estimation is a cautious approach.

Strengths:

a.) The CRI is simple, and hence other scientists can repeat its analyses, as evidenced by the fact that others have adopted the matrix approach and conducted their own estimates of lambda.

b.) The CRI emphasizes documentation of data and data sources, and places a greater premium on presenting basic data and hypothesis testing using data prior to modeling.

c.) The CRI is attempting to deal with observation error by designing estimation procedures that are robust despite its presence, and by focusing on obtaining direct estimates of its magnitude. It is unclear that any analytical framework can be reliable if it does not address sampling error in some fashion.

d.) The CRI is extremely responsive to new and/or updated data.

e.) The CRI places emphasis on transparency and presentation so that others may understand its methods. One manifestation of this is the CRI's emphasis on publishing in peer-reviewed journals.

5. What are the assumptions of your model?

Because the CRI is such a simple quantitative analysis there are not many assumptions. However two critical simplifications that the CRI has made to date need to be mentioned.

a. The first simplification is the assumption of a lack of density-dependence in current dynamics. Previous models applied to salmon in the region have assumed a Ricker form of density-dependence. Our analyses indicate that the statistical evidence to support such density-dependence is absent (the density-dependence we occasionally do detect takes the form of density-depensation, or declining productivity as abundance declines).

b. The second critical assumption has been to treat variability in recruits per spawner as lognormal variation from a temporally homogeneous stochastic process (i.e., it assumes there are no directional trends underlying the variation).

Both assumptions, and especially how one tests them with time series of data, are inter-related. In McClure et al (2000), the CRI reports results from likelihood ratio tests for density dependence in the time series of 69 stocks, and finds strict density-dependence in only eight cases. In all of these cases there is the suggestion of declining recruits per spawner with declining numbers, or density-depensation (which is not the form of density-dependence used in the Ricker formulation). Unfortunately, it is very hard to distinguish between a hypothesis of density-depensation and a temporal trend. The CRI is developing methods based on information theory to tease apart these two mechanisms.

6. How is uncertainty addressed?

Uncertainty is dealt with in several ways. First confidence intervals for estimates of annual population growth are calculated. Second, when we have almost no data, as is the case for the reproductive fitness of hatchery fish, we estimate lambdas using an upper and lower bound for hatchery fish fitness (i.e., we assume the fitness of hatchery fish is zero at its lowest or equal to that of wild fish at its highest). In other cases, we perform numerical calculations over a wide range of potential values. For example, we calculate the expected increase in lambda for Snake River spring/summer chinook salmon over a broad range of potential improvements in estuarine/early ocean survival due to dam breaching (ranging from 0% to 180%, in 20% increments; see Figure VI-5). A third way of addressing uncertainty is to use multiple methods to calculate key parameters. For example, we calculate sensitivity measures (change in lambda associated with a 10% reduction in mortality) off baseline matrices that are derived in several different ways e.g., using different time horizons (beginning in either 1980 or 1990) or using different methods to derive matrix parameters (using SAR's to back-calculate estuarine survival, or starting with a literature value of 7% for estuarine survival). Lastly, we encourage actual plots of data, such as mean recruits per spawner as a function of subwatershed habitat (see figure extracted from Regetz manuscript in section V of McClure et al 2000).

7. Why do we think our model's predictions are accurate?

We do not always think the inferences from our analyses are accurate – it all depends on which inferences? (or which predictions?).

Examples of "accurate" predictions are: the conclusion that even if downstream survival were increased to 100%, spring/summer chinook lambdas remain substantially lower than 1.0 and the populations are still at severe risk. After the fact, this is an obvious point. But as a numerical result it puts much of the focus on the details of downstream passage in perspective (suggesting that the emphasis of passage research now should turn to consequences for survival below Bonneville dam). Another "accurate" inference is the relative opportunity for improving lambda by acting on first year survival and estuarine/early ocean survival relative to survival in other lifestages. At the ESU-level across the entire Columbia Basin, our prediction that major harvest reductions could achieve the needed improvements in lambda for Upper Willamette Chinook, Lower Columbia Chinook, and the Snake River fall chinook is likely accurate.

Other predictions or estimates are only moderately accurate. For example our estimates of lambda (annual population growth rates) are probably the best available summary of the current trends in populations, subject to the uncertainty about how to adjust the estimates for the presence of hatchery fish on spawning grounds.

Finally, there are aspects of our analyses whose accuracy needs further study. The best example concerns "probabilities of extinction". We do not require that probabilities of extinction are accurate, but we do require that their relative ranking of risks among stocks or ESUs is accurate. We have launched several analyses to determine how well relative rankings of extinction risk stand up in the face of sampling error and fundamental violations of the underlying model.

Lastly, speaking about models strictly in terms of "accuracy" does not do modeling justice. We can learn from models quite independent of accuracy. A good example from recent CRI analyses concerns what we have learned about the need to better resolve hatchery inputs on wild spawner grounds. We have learned that it is foolish to take seriously absolute predictions of extinction risk (or population growth) in the absence of rigorous samplings of marked hatchery fish on spawning grounds and empirical evaluations of the reproductive rate of those hatchery fish relative to wild fish. Without modeling, this result is evident to some – but the exercise of estimating lambda under different assumptions about the hatchery contribution makes the importance of this uncertainty dramatically evident. This is NOT an uncertainty that other analytical frameworks have drawn attention to.

8. How does your modeling effort relate to or contrast with the other three modeling efforts?

The distinguishing features of the CRI are:

a. The CRI is the analytical approach that sticks closest to the actual data.

b. Unlike the other analytical approaches, it is possible to readily grasp the analyses undertaken by the CRI, and redo the calculations under slightly modified assumptions.

c. The CRI examines eleven different ESUs in the Columbia Basin – none of the other models tackle this biological and geographic scope.

d. The CRI focuses first and foremost on annual rate of population change (adjusted for environmental variability), and the risk of extinction.

PATH is the only other analytical approach that starts with spawner counts as the key input data and uses transient population dynamics as a key response variable. But unlike CRI, PATH has produced analyses for only two of the Columbia Basin ESUs and tends to focus on hydropower. EDT is a broader analysis than is CRI in terms of its willingness to evaluate alternative management scenarios. However, EDT is primarily an expert system. Although useful for generating hypotheses and summarizing spatially explicit habitat descriptors, EDT does not incorporate much about fish population dynamics. To see this, simply recognize the fact that the most recent four years of Snake River Spring/Summer Chinook recruits per spawner data have no influence on EDT's answers. We do not know enough about ICBEMP to comment on it, except to mention that it seems to be primarily a habitat-focused analysis. It is not clear that ICBEMP's framework could ever allow an evaluation of harvest or hatchery impacts.

9. What advice would you give decision-makers on how they should use your model to support decisions regarding salmon recovery in the Columbia River basin?

Below is an excerpt from the EXECUTIVE SUMMARY of McClure et al:

Generally, ESUs or stocks with the most rapid rates of decline – the lowest lambdas -- require the most improvement to mitigate extinction risks. However, this generality is complicated by the fact that low populations and high environmental variability can exacerbate extinction risks beyond what might be expected from lambda alone. The magnitude of improvements required in lambda ranged from less than 1% to as much as 65%, with most values falling between 5% and 20%.

The more difficult task is exploring opportunities for improving lambda, i.e., increasing the number of recruits per spawner. The well-known "four H's" (hydropower, habitat, hatcheries and harvest) represent the human-influenced arenas in which management can be altered in hopes of recovering ESUs. But because these four H's vary enormously in the areas occupied by different ESUs, it is unlikely that a simple prescription can be drawn up that fits all ESUs. For example, the number of dams per kilometer varies from 0.4 to 2.8 depending on the region associated with each ESU. Land use characterization also varies widely across regions occupied by ESUs, with some regions characterized by a high percentage of rangeland (Upper Columbia and Snake Rivers), urbanization (lower Columbia, and upper Willamette Rivers), or cropland (upper Willamette River). At the finer scale of index stocks, preliminary analyses indicate that three habitat variables at the subwatershed scale explain 60% of the variation in recruits per spawner: (1) percent of land classified as urban, (2) proportion of stream length failing to meet EPA water-quality standards, and (3) the ability of streams to recover from sediment flow events. Lastly, although nearly 100 hatchery facilities in the Columbia Basin release approximately 150 million smolts annually, the magnitude of this hatchery production varies by an order of magnitude among ESUs. The impact of this hatchery production is difficult to analyze because of the lack of large-scale controlled experiments. Some preliminary analyses suggests that in "poor ocean years" hatchery fish compete with wild fish and lower the survival rates of the wild fish.

Rates of population decline and extinction risks vary widely across the Columbia River Basin, suggesting that management needs vary in accord with these different levels of risk. Most imperiled are Upper Columbia Spring Chinook, Middle Columbia steelhead, Upper Columbia steelhead, and Upper Willamette steelhead ESUs.

The amount of improvement in recruits per spawner that is required to mitigate risks can be modest (less than 1%) or quite large (as high as 65%). When needed improvements are modest there are probably management options, but when needed improvements are large there is little room to be selective about what actions are taken. We must do everything possible to increase recruits per spawners before it is too late. A lambda of 0.9 means that in less than 7 years a population is likely to be reduced to half its current level.

Reductions of harvest represent an easily identified mechanism for improving recruits per spawner in a few ESUs (Lower Columbia Chinook, Upper Willamette Chinook, and Snake River Fall Chinook). In other ESUs we lack data for making confident quantitative predictions about the likely effects of any particular management action. This is even the case for the much-studied Snake River Spring/Summer Chinook salmon where risks are substantial and the need for action is striking (particularly if one factors in the recent declining trend in recruits per spawner). Although there is some evidence that dam breaching is necessary for mitigating the extinction risk faced by Snake River Spring/Summer Chinook salmon (especially given the lack of evidence that needed improvements can be made by non-breaching management actions), it is highly unlikely that dam breaching alone will recover these populations. Hence, even in this most-studied of all cases, actions will be predicated on uncertainty. But what is not uncertain is the substantial rates of decline for Snake River Spring/Summer Chinook salmon and even worse rates of decline for several other ESUs.

In summary, the scientific uncertainty surrounding the likely outcome of everything but harvest reductions is not an argument for inaction, especially given the high risks faced by several ESUs. Quite the contrary. This level of uncertainty is, however, an observation that the public and policy makers should be aware of. From a scientific viewpoint the ideal action is rapid, targeted management action with effective monitoring programs. Secondly, establishment of quantitative links between management actions and salmon productivity are obviously a priority area for research. The region has suffered from an inattention to standardized reporting of data and analyses at a large scale and as a result currently lacks the scientific information required to make quantitative assessments of management scenarios. It is imperative that this last point be emphasized to the public and policy makers: collectively we have failed to manage Columbia River Basin salmonid populations and are now forced to undertake management actions as experiments, accepting that some will fail, but if they are properly designed, we can learn from our mistakes.

In general, we feel that decision-makers might be well-served by drawing on all of the available analytical tools, either in their existing or modified form, and not just on CRI. First, it is worth noting that EDT and CRI are not “dueling” models; in fact, they attempt to predict totally different response variables. CRI examines annual rates of population growth and extinction risk, neither of which are EDT inputs or outputs. However EDT does identify “good habitat conditions” versus “bad conditions”, and assesses responses to changes. Both of these represent hypotheses that could be tested using the machinery of CRI. In addition, the spatial data synthesized by EDT might be used in empirical models aimed at testing relationships between habitat and salmonid productivity as measured by demographic or spawner data. Second, although PATH and CRI are in one sense at odds with one another because of contrasting modeling and scientific philosophies, even these two approaches could be profitably combined, or minimally learn from one another. In fact, already CRI and PATH are learning from one another (in spite of what often seems like an acrimonious scientific debate). For example, CRI has altered the way it parameterizes its Snake River matrices, and now uses PATH SAR estimates as the starting point with which to solve for first year survival. Conversely, PATH has begun to use a simplified matrix formulation to explore its ideas about extra mortality, and has added calculation of extinction probability to its repertoire of output statistics. Clearly CRI and PATH will never "agree" in the sense of approaching analyses in the same manner – however, if they can maintain a scientific dialogue, there is some opportunity for each approach to be improved. The main obstacles to a reconciliation of PATH and CRI are that (i) the two approaches operate on different timescales (the CRI seeks results in months, whereas PATH seeks results in years), and (ii) once one steps beyond Snake River chinook salmon, the detailed and complicated models typical of PATH clearly cannot be supported. We do not know enough about ICBEMP to discuss how its approach may or may not complement CRI.

PATH Answers Part I - General and FLUSH

Paul Wilson, CBFWA

Question 1.

PATH grew out of previous efforts by operating agencies and state, tribal, and federal fisheries agencies to compare and improve models used to evaluate management options intended to lead to recovery of listed salmon and steelhead stocks. The Plan for Analyzing and Testing Hypotheses (PATH) is a formal and rigorous program of formulating and testing hypotheses. It is intended to identify, address and to reduce uncertainties in the fundamental biological issues surrounding recovery of endangered spring/summer chinook, fall chinook, steelhead and sockeye stocks in the Columbia River Basin.

The objectives of PATH are to:

a) determine the overall level of support for key alternative hypotheses from existing information, and propose other hypotheses and/or model improvements that are more consistent with these data (retrospective analyses);

b) assess the ability to distinguish among competing hypotheses from future information;

c) advise institutions on research, monitoring and adaptive management experiments that would maximize learning; and

d) advise regulatory agencies on management actions to restore endangered salmon stocks to self-sustaining levels of abundance (prospective and decision analyses).

Question 2.

PATH analyses have focussed on Snake River ESA species. Species outside of the Snake R. basin were not addressed in detail in PATH until very recently, so conclusions presented here apply to Snake River ESA species only.

Retrospective Analysis (Spring/Summer Chinook Salmon Only)

1. Declines in numbers of spawning salmon exhibited population-specific patterns.

2. Declines were greater for Snake River Basin and upper Columbia River Basin populations than for lower Columbia River Basin populations.

3. Declines in Snake River Basin populations after 1974 were greater than declines before 1974, and greater than declines in lower Columbia River Basin populations after 1974.

4. Differences in survival and productivity between Snake River Basin populations and lower Columbia River Basin populations coincide in space and time with development of the Hydropower System.

5. The significant decline in Snake River Basin populations after 1974 does not coincide with habitat degradation. Degradation in habitat occurred mostly prior to 1974 and spawning and rearing habitat for some of the Snake River Basin populations has remained in good or pristine condition.

6. The degree to which artificial propagation contributed significantly to declines in survival of listed Snake River Basin populations is uncertain.

7. Declines in survival of Snake River Basin populations occurred at the same time as substantial declines in harvest rates. These trends are contrary to the assumption that harvests are significant contributors to declines in survival of Snake River Basin populations after 1974.

8. Different salmon populations appeared to respond differently to changes in climate. However, no climatic index could explain differences between declines in survival of Snake River Basin and lower Columbia River Basin populations.

9. All models suggest that mortality of Snake River Basin populations in the Columbia River downstream from Bonneville Dam and in the ocean (extra mortality) is greater for "transported" fish than for "in-river" fish that migrated past dams and through reservoirs. The efficacy of transportation is expressed as the "D-value", which is a reflection of the ratio of extra mortality of "transported" fish to extra mortality of "in-river" fish. The retrospective D values have been estimated to be less than one (0.63 to 0.34).

Prospective Analysis and Risk Assessment (Spring/Summer Chinook Salmon and Steelhead, and Sockeye Salmon)

1. The hydropower system options examined capture three general approaches for operating and/or configuring the hydropower system in the future.

a. Three options use barges to move juvenile fish around dams and reservoirs. One option (A1) describes the 1995 BIOP. The other two options (A2/A2') describe improved transportation under conservative and liberal assumptions about the extent and effectiveness of improvements.
b. Three options are called "natural-river". Two options (A3-3yr and A3-8yr) assume Snake River dams are removed after 3 years and 8 years, respectively. The other option (B1) also draws the level of John Day Reservoir down. Both A3 options were explored for spring, summer and fall chinook. Two options (A6/A6') improve existing passage facilities at dams and use flow augmentation to speed migration. These options are called "optimum in-river migration".

2. The "optimum in-river migration" options (A6/A6') were not analyzed in the same way as the "transportation" and "natural-river" options. Instead of using a formal quantitative risk analysis, the "optimum in-river migration" options were analyzed relative to the "transportation" and "natural-river" options. This was necessary because time constraints prevented hydrologists from developing the "optimum in-river migration" options in sufficient detail for the quantitative analysis. Results showed that listed salmon were no more likely to survive and recover under A6 than A2.

3. Of 14 uncertainties examined, seven proved to most influence estimates of survival and recovery under each hydropower system option. Of these seven, the most influential assumptions were those included in the passage/transportation models and those describing the mortality of salmon after they have passed through the hydropower system. These assumptions most influenced the results because they accounted for the largest proportion of the mortality salmon experience over their life cycle.

4. For spring and summer chinook salmon, two of the "natural-river" options (A3-3yr and B1) exceeded all of the standards. A3-8yr met the 100-year survival and 48-year recovery standards, but missed the 24-year survival standard by less than 1 percentage point. This was also true for the weighted results. None of the "transportation" options met either the 24-year survival standard or the 48-year recovery standard under equal weights. This was also true for the weighted results.

5. For spring and summer chinook salmon, the "natural-river" options had higher probabilities of achieving the survival and recovery standards than the "transportation" options. "Natural-river" options met the standards over a wide range of assumptions. In fact, the "natural-river" options met the 100-year survival and the 48-year recovery standards under the most pessimistic set of assumptions. "Natural-river" options were also less risky than the "transportation" options. The projections had relatively little variability over the full range of assumptions.

6. For steelhead, a hydropower system option is likely to meet survival and recovery standards if it meets the standards for Snake River spring and summer chinook salmon. However, our analyses did not address whether the survival and recovery standards for steelhead would be met if a hydropower system option fails to result in an acceptable likelihood of survival and recovery for spring and summer chinook salmon.

For sockeye salmon, recovery is less likely than for Snake River spring and summer chinook salmon under "transportation" options, if high rates of descaling, which appear to be associated with bypass screens, are a primary source of injury and mortality. This is because "transportation" options rely on bypass systems to collect juvenile sockeye salmon. The likelihood of recovery of sockeye salmon under "natural-river" options, relative to other species, was not analyzed.

Fall Chinook

1. Models that include passage model inputs fit the historical time series of Snake River spawner-recruit information much better than simple models that only consider inherent stock productivity. However, models of intermediate complexity that do not include passage model inputs provide better fits than most models with passage model input.

2. The best fits to the spawner-recruit data are obtained when D is estimated from the spawner-recruit information, or when D is fixed below a value of 0.2. Models that fix D at values ≥ 0.2 have a poorer fit to the spawner-recruit data, and models that assume D=1 have a much poorer fit. Estimates of D and spawning effectiveness are related because hatchery production and transportation of fall chinook began at around the same time in the historical period.

3. Models that fix hatchery spawner effectiveness at a high value (0.7 or 1.0) provide better fits to the historical spawner-recruit data than models that estimate E. However, models that fix hatchery spawner effectiveness at low values (0.0) provide worse fits to these data than models that estimate E. Estimated E values are generally around 0.7 to 1.0, unless the D value is fixed at a high value.

4. Including a common ocean effect that affects several Columbia River fall chinook stocks results in a strongly better fit over models that do not include this effect. The best estimates of these common effects are based on data from the Snake River and Deschutes stocks.

5. All hydrosystem actions project an improvement in survival rates and in spawner abundance because of assumptions about system operations built into the management scenarios, such as flow augmentation at levels prescribed by the 1995 Biological Opinion.

6. Projected outcomes of actions depend strongly on what is assumed about D (the estuary/ocean survival rate of transported fish, relative to the estuary/ocean survival of non-transported fish).

7. All hydrosystem actions meet survival standards (probabilities of exceeding survival escapement thresholds are greater than 0.7), regardless of what is assumed about the estuary/ocean survival rate of transported fish.

8. All drawdown actions meet recovery standards (probabilities of exceeding recovery escapement thresholds are greater than 0.5), regardless of what is assumed about the estuary/ocean survival rate of transported fish. The drawdown actions (A3, B1) exhibited the most robust response across those uncertainties considered to date, and produced higher recovery probabilities (as well as higher average spawning escapements) than other actions.

9. There are non-breaching actions (actions that do not involve drawdown of dams) that meet recovery standards, although there is no single non-breaching option that meets recovery standards under all assumptions about the relative survival of transported fish. If transported fish are assumed to have high relative survival (i.e. high D), maximizing transportation will achieve recovery standards. If transported fish are assumed to have low relative survival (i.e. low D), then retaining current system configuration and allowing all smolts to migrate in-river achieves the recovery standards.

10. In addition to transport survival assumptions, model results are sensitive to alternative ocean and in-river harvest rate targets, alternative survival and recovery thresholds, and (under the highest D assumption) alternative assumptions about upstream survival rates of adults.

Sensitivity Analysis (Spring/Summer and Fall Chinook)

1. Changing assumptions about the quality and quantity of freshwater spawning and rearing habitat had minor influences on estimates of the probability of meeting survival and recovery standards for spring and summer chinook salmon and did not affect overall ranking of hydropower system options. Alternative assumptions about FGE, conversion rates following drawdown, and effectiveness of the predator removal program generally had small impacts on the probability of meeting survival and recovery goals for both spring/summer and fall chinook.

2. Small reductions in already small spring/summer harvest rates had minimal effects on the probability of meeting survival and recovery standards. These improvements in the probability of meeting survival and recovery standards under “transportation” options were not sufficient to change the ranking of hydropower system options. “Natural-river” options still produced higher probabilities of meeting survival and recovery standards than “transportation” options.

3. Small changes in ocean and in-river harvest of fall chinook salmon had minor effects on the probability of meeting 48-year recovery standards for “transportation” options. These changes in the probability of meeting recovery standards under “transportation” options were not sufficient to change the ranking of hydropower system options. Large changes (e.g. reducing in-river harvest by 50% and ocean harvest by 50 or 75%) had a significant effect on probability of meeting recovery standard. However, the standard was met with transport options only for one ‘D’ hypothesis examined, and “Natural-river” options still produced higher probabilities of meeting survival and recovery standards than “transportation” options.

4. Explicitly incorporating additional mortality, i.e. from sources that may not be reflected in the historical spawner and recruit data up to brood year 1990, into spring and summer chinook salmon analyses affected all hydropower system options equally, and thus did not change their ranking. These additional sources of mortality

approximated recent levels of predation on salmon smolts by Caspian terns and other bird predators in the estuary. Incorporating the additional mortality caused all the hydropower system options to miss the 24-year survival standard. It also caused the "transportation" options to miss the 100-year survival standard and 48-year recovery standard.

5. Increases in the survival rates of adult spring, summer, and fall chinook salmon during their upstream migration after drawdown of John Day Dam had minimal effects on overall results.

Overall Conclusions:
Prospective and Risk Analyses

1. The "natural-river" options exceeded all three standards NMFS uses to determine jeopardy for spring, summer, and fall chinook salmon, with one exception. The likelihood of survival of spring and summer chinook salmon missed the 24-year survival standard by less than one percentage point when drawdown of Snake River dams was delayed for eight years. In most cases, the "natural-river" options also met the standards under the most pessimistic assumptions. None of the "transportation" options met the recovery standard, except under very optimistic assumptions.

2. Weighting certain key assumptions to reflect recommendations by the Scientific Review Panel did not significantly change the results. "Natural-river" options still met all the standards, while "transportation" options failed to meet the recovery standard.

3. "Natural-river" options were less risky than "transportation" options for spring/summer and fall chinook salmon. The projected likelihood of meeting the jeopardy standards under the "natural-river" options was least variable over the full range of assumptions. See Marmorek et al. (1998)

Sensitivity Analyses

1. Changing assumptions about the quality and quantity of freshwater spawning and rearing habitat, harvest rates, and survival rates of adult salmon to their birthplace had minor influences on estimates of the probability of meeting survival and recovery standards for spring, summer and fall chinook salmon and did not affect overall ranking of hydropower system options.

2. Explicitly incorporating additional mortality into spring and summer chinook salmon analyses equally reduced the likelihood of all hydropower system options meeting the standards, and thus did not change their ranking. Incorporating the additional mortality caused all of the hydropower system options to miss the 24-year survival standard. It also caused the "transportation" options to miss the 100-year survival standard and 48-year recovery standard.

Questions 3, 5, and 6.

Kinds of data used in PATH modeling:

Run reconstruction data:

Redd counts from index areas; dam counts of adults; harvest rate estimates from CWT recoveries; carcass scale data (for spawner age estimation); hatchery brood removal estimates; hatchery-origin natural spawner estimates.

Passage modeling:

Physical Data. Both the CRiSP and FLUSH passage models use specific flow rate, reservoir elevation, spill rate, and temperature (for FLUSH, fall chinook only) data in their passage models. These variables influence several mechanisms within in the models such as fish travel times, relative usage of dam passage routes, and predation rates.

Biological Data. Data on Initial Emigration Timing (CRiSP and Fall FLUSH only); fish travel time estimates (from PIT-tags and coded-wire tags); reach survival estimates (from NMFS, PIT-tags and CWTs); predator abundance and consumption data, survival estimates for different dam passage routes (spill, turbine, bypass); spill effectiveness estimates; fish guidance efficiency estimates; passage indices at dams; smolt descaling data.

Other life stage data used:

Subbasin specific hatchery release data; juvenile dam counts; SAR estimates from CWT and PIT-tag data (of both transported and non-transported fish); radio tag data for adult upstream migration and pre-spawning mortality; climate data (e.g., sea surface temperature, upwelling/downwelling transition dates, drought indices); land use data (e.g. percent watershed logged).

Assumptions/Uncertainties:

Uncertainties in past conditions due to incomplete data and potentially confounding influences generate a range of alternative assumptions about historical conditions, such as the mortality of fish at specific dams in past years, or the success of past transportation experiments. These alternative assumptions about the past, together with historical flow information, are used in retrospective modeling analyses that generate quantitative estimates of parameters needed to run models into the future. This requires running both passage models, which estimate survival from Lower Granite Reservoir to Bonneville Dam and life cycle models. Spawner-recruit data and environmental data (e.g., climate indicators) are used for calibration of the life cycle models' stock production functions and other parameters. The retrospective modeling analysis quantifies our understanding of the variability in survival rates, and the factors that affect them. Results from the retrospective analysis are passed to the prospective analysis. The prospective modeling analysis quantifies the range of possible futures, expressed as specific performance measures. This set of possible futures depends on:

• the understanding and estimated parameter values gleaned from the retrospective analysis;
• the specific future action under consideration (scenarios A1, A2, A2', A3, B1). This set of actions has been developed by the Implementation Team (I.T.), and draws from previous experience of analyzing a much larger set of options (refs: Biological Opinion; System Operating Review; System Configuration Study).
• the expected flows associated with each action; and
• assumptions about future conditions, including passage survival assumptions such as fish guidance efficiency through bypasses around dams, and non-passage assumptions such as harvest schedules, habitat improvements and future climate.

For the prospective analysis, the alternative hydrosystem management actions are evaluated by simulating their consequences using a linked set of models in a four-step process to generate performance measures:

1. A hydro-regulation model translates each management option into the mean monthly flows which would be observed in the Snake and Columbia Rivers at various locations, (the U.S. Army Corps' HYSER model has been used for the scenarios included in this report). The hydro-regulation model is run for the water years 1929-1988 to generate a representative set of flows, and this information is used as input to the passage models.

2. A passage model translates the projected set of flows, spills, and dam configurations and operations for a given year into the estimated passage survival of both transported and non-transported smolts through the migration corridor from the head of Lower Granite Reservoir to the tail-race of Bonneville Dam. The passage models simulate passage survival rates under each management action for a given set of water years, to compute the improvement in survival relative to the retrospective period. The longer term water record (i.e., 1929-1988) is considered in step 4. We have used two different passage models, CRiSP and FLUSH, which use different approaches to predicting passage survival rates.

3. One of the key pieces of information passed from the retrospective modeling analysis to the prospective analysis are estimates of the ratio of post-Bonneville survival rates of transported to that of non-transported fish. These ratios are generated by combining estimates of historical passage survival rates with the results of transportation experiments.

4. A life-cycle model (BSM) generates a range of possible spawner abundances for each stock and year, under each management option. It does this by combining information produced by the passage models (i.e., the projected passage survivals, fraction of fish transported, and post-Bonneville survival assumptions) together with estimates of the

other (non-passage) influences on survival (i.e., stock productivity, adult survival during upstream migration and harvest, post-Bonneville mortality, climate conditions, and habitat changes). The life-cycle model performs a thousand simulations for a given set of passage model inputs to ensure that the full range of possible ways the system works, and thus the full range of possible futures, is adequately simulated, and that the uncertainty in performance measures is properly estimated. These simulations randomly select passage model outputs from each of the given set of water years according to how frequently the flow in each kind of year occurred in the long term historical record (1929-1992). For example, an extremely low flow year like 1977 (the lowest flow in the entire 1929-1992 period) is selected much less frequently than a more typical flow year like 1979 or 1985. For spring/summer chinook, the life cycle model also considers alternative assumptions with respect to whether upstream and downstream stocks have some common responses to climate fluctuations (DELTA approach) or respond independently (ALPHA approach).

The decision analysis approach (of which the Bayesian life cycle model is only one component) allowed the analysis to consider all possible combinations of all relevant uncertainties. The Bayesian life cycle simulation model (BSM) was essential to carry uncertainties in climate and stock-production relationships through the analysis. BSM was modified to allow for the possibility that upstream and downstream stocks have different estuarine and ocean survival – this took extra time and made the analysis more complex, but was an important alternative to the hypothesis of common year effects.

Question 7

The MLE (Maximum Likelihood Estimation) life cycle model, which uses retrospective data to generate information used in the prospective life-cycle model (BSM), was developed by an independent scientist (R. Deriso) in consultation with the other PATH participants. The procedures used estimating parameters ultimately used in the analysis of actions in the hydrosystem and other H’s were analyzed for sensitivity to such things as spawner measurement error and different parameterizations. Biases arising from using the wrong set of assumptions were tested for by generating simulated recruitment data and assumed passage mortality, and then using the MLE model to estimate the parameters (whose ‘actual’ values came from a distribution with specified mean and variance). Lowest bias models found from this procedure were the ones used in the decision analysis. The median predicted recruitment estimated by the MLE for each brood year can be compared to the observed recruitment for each index stock [slide].

The life-cycle models were used not to make point estimates of future spawner abundances, but to generate statistical distributions of these abundances. Any one run (i.e., a setting with particular inputs and parameter distributions reflecting, e.g. a particular passage model and extra mortality hypothesis) results in a distribution of possible outcomes. The accuracy of the distribution produced by any one run of the BSM depends on the accuracy of the hypotheses that run relies on (as well as the error inherent in the life-cycle model configuration). Use of historical information on recruitment and juvenile passage survival helped constrain the analyses of prospective actions to plausible outcomes, although intersection of some hypotheses resulted in highly optimistic outcomes. The influence of these highly optimistic runs on the overall conclusions was minimized somewhat when SRP weightings were applied to the run results to generate overall distributions.

Actions were categorized by assessing the performance of the stocks over a range of combinations of likely hypotheses, in order to determine the ranking of actions with respect to the performance standards (using either equal weighting of hypotheses or SRP weighting). The documentation tried to emphasize that relative ranking of actions may be more important than information on performance of an action relative to absolute standards, given the knife-edge nature of the performance standards. Rank order of actions was insensitive to choice of life-cycle model configuration (alpha or delta), though in general delta was more optimistic with respect to performance standards.

The passage models used (especially for spring/summer chinook) represented very different approaches and predicted substantially different pictures of the relative benefits of transportation-based and drawdown scenarios. Drawdown scenarios were better than transportation-based scenarios under almost all assumptions (spring/summer chinook) or most assumptions (for fall chinook) for both models; the degree to which they were better depended on passage model. The competing passage models and the different hypotheses under which each was run encompass the likely outcomes of each hydrosystem alternative. The ability of the passage models to predict reach survival was judged by producing both 'retrospective' and 'prospective' model estimates to compare to reach survival estimates [transparency]. The prospective-type predictions are particularly useful because information that would not be available in advance of a migration season is excluded from the prediction method. Extensive diagnostic output was produced so the mechanisms and hypotheses of the models were explicitly presented to SRP and others for comparison with empirical estimates and with rationale for model hypotheses. More important, passage model output used in conjunction with transportation effectiveness estimates in life-cycle models to compare predicted SAR and R/S to observed values using multiple methods for estimating goodness of fit.

Question 8

PATH is an open, inclusive, fully collaborative, independently facilitated project. PATH gets its priorities from an inter-agency policy team. It makes use of expertise in several areas: fisheries biology; analytical methods (including modeling and statistics); hydrosystem operations; familiarity with the quality of empirical information (e.g. estimates of escapement, passage survival, smolt to adult survival, dam survival, barge survival); conservation biology and decision analysis. PATH has a healthy balance of expertise in these areas. The proof of this assertion lies in the PATH Weight of Evidence report and associated submissions, which integrated together all of these different types of expertise, in a balanced synthesis (see independent scientists' and SRP scientists' comments on the Weight of Evidence report, attached to this letter). Incidentally, PATH does not "vote" on conclusions or hypotheses; the hypotheses, analyses and evidence that carry the most weight are those which most strongly survive internal and external peer review.

Unlike the other efforts, PATH has concentrated on Snake River stocks. Analyses of other Columbia stocks have served primarily to aid in the Snake River decision analysis. The concentration on Snake River stocks has allowed PATH to perform a most comprehensive analysis of two of the stocks (spring/summer chinook and fall chinook). PATH analyses focussed on data-rich stocks. "The PATH decision analysis is by far the most comprehensive example of applied decision analysis for any resource management problem that I know of. It is an extremely complex undertaking, but if anyone can suggest a better approach, I'd like to hear about it" (Randall Peterman). PATH's explicit inclusion of uncertainty and variability is unprecedented in the region—it did not rely on point estimates of vital rates when comparing management actions, for instance. PATH considered all the H's, but its charge was to help make a decision on long-term hydrosystem configuration. PATH performance measures were primarily those of NMFS's jeopardy standard, and did not attempt to estimate absolute extinction probabilities. PATH used historical data on productivity of target stocks and downstream stocks to put management efforts in context and to help estimate their likely impacts.

CRI has a much broader mandate. CRI is a non-collaborative, NMFS process, with input from other entities in the form of comments on CRI analyses. CRI scientists have a broad background in conservation biology, but limited experience working with anadromous salmonids, and many are new to the region. CRI is charged with analyzing all ESA listed anadromous stocks in the region, regardless of the amount or quality of data available. A major objective has been consistency of approach to stocks in different river basins and with vastly different quality and quantity of data available. Simplicity and transparency are stated to be important goals of the process.

CRI has used some data or results developed in PATH. However, in the CRI analyses of Snake River chinook stocks, inclusion of alternative hypotheses, such as experience at one life stage affecting survival probability at a later stage, has been cursory. Point estimates of vital rates have been used to compare the potential benefits of different management actions. Two types of models have been used: a "Dennis-type" extinction model to estimate likelihood of extinction in a given time, assuming that conditions do not change from the 1980 to early or mid-90s period; and a deterministic Leslie matrix model to estimate expected changes in the mean growth rate under the assumption of a fixed amount of mortality being reduced in individual life stages. Data prior to 1980 are not used, and no context has been provided to quantify the likelihood of actions in different life stages actually being effective at improving the population growth rate. CRI's main analysis was a "baseline" scenario, which assumed no improvement over recent conditions. PATH did not analyze a corresponding scenario, since current operations (A1) were assumed to represent improvement over pre-1995 conditions.

EDT is a subbasin-focused modeling system. Environmental attributes of different watersheds are described and quantified. EDT is primarily a single-entity process. EDT also has a broader agenda than PATH, attempting to analyze the entire Columbia River basin. It is broader than CRI in that it includes analysis of resident fish and wildlife species, and not just ESA-listed anadromous species. The inclusion of expert opinion is more extensive, but less formal than the processes used in PATH. The performance measures are less sharply defined than in PATH or CRI.

Question 4

Strengths

- PATH is a cooperative, multi-agency process, representing a broad array of viewpoints.
- Internal review and debate lends focus to arguments and hypotheses.
- All PATH work products are peer-reviewed by external scientists.
- The process includes a neutral facilitator and a number of independent scientists, including experts on decision analysis and elicitation of expert opinion.
- Two different life cycle models. First model used year (climate) effect estimated using downstream stocks; second modeled stocks in each region separately. First prospective model used relative improvement (ratio of prospective to retrospective survival estimates) in downstream passage and extra mortality hypotheses; second model plugged in actual prospective estimates to model prospectively.
- Explicitly included natural environmental variation and human-induced variation in vital rates.
- Did not attempt to model absolute extinction.
- Differing opinions on key hypotheses, even after weight of evidence. Decision analysis approach of PATH made it unnecessary to have complete agreement to determine which actions had best chance of achieving recovery over range of uncertainty.

Weaknesses

- PATH is a multi-agency process: it takes time to reach agreement and/or consensus. Sometimes difficult to meet internal and external deadlines.
- Resources (key personnel) are limited.
- There will always be limited data for building models, testing hypotheses about the past, and testing hypotheses about the future.
- Factors not modeled include: behavioral, productivity and genetic interactions between populations; the effects of the quantity and quality of mainstem river habitat; hydrosystem impacts on the estuary.
- Did not attempt to model absolute extinction.
- Analyzes intrinsically complex natural resource problems in great detail. Not transparent to casual observer. Difficult for most to read all of the documents.
- Minority opinions can obscure robustness of primary findings.

Question 9.

I would advise decision-makers to keep in mind that PATH was conceived of as a preponderance-of-evidence approach from the start; there was never any possibility of arriving at consensus on one action that, if implemented, could be guaranteed, beyond all reasonable doubt, to recover the listed stocks. The decision analysis could show only under which assumptions different outcomes were achieved for each analyzed action, and also which actions were most likely to benefit the stocks under the widest range of assumptions reflecting uncertainty in biological processes. PATH was a fully collaborative process; NMFS and other federal agencies were intimately involved. It was *the* collaborative process initiated by the region to develop quantitative biological information to aid in making the 1999 decision on long-term operation and configuration of the hydrosystem.

No other process has been so fully and thoroughly peer-reviewed, and no other process has considered as much of the relevant information. Further, the results attained in PATH are **not** dependent on the particular configuration of models used in PATH: analyses using the same data but using the Leslie matrix model advocated by CRI give strikingly similar results about the relative effectiveness of the proposed FCRPS alternatives in preventing extinction and achieving recovery of listed Snake River stocks (Oosterhout et al. 2000). The interagency Drawdown Regional Economic Workgroup (DREW), which was charged with estimating the costs and benefits of the alternative FCRPS actions, used PATH to provide information on biological outcomes of the different actions.

References

Marmorek, D.R., C.N. Peters and I. Parnell (eds.) 1998. PATH final report for fiscal year 1998. Prepared by ESSA Technologies Ltd., Vancouver, BC, 263 pp.

Oosterhout, G. and 8 others. 2000. A technical review of the National Marine Fisheries Service Leslie matrix model of Snake River spring and summer chinook populations. April 28, 2000.

PATH Answers Part II - CRiSP and Alpha Life Cycle Model

Jim Anderson, University of Washington
Phone: 206-543-4772
e-mail: jim@cbr.washington.edu

Date: March 20, 2000

1. What is the purpose of your modeling effort? What questions or problems were your models designed to address?

The purpose of PATH was to evaluate the effectiveness of a limited number of Snake River salmon recovery actions including breaching the lower four Snake River dams, increased transportation, and reduced harvest. Hatcheries and habitat were evaluated in a cursory manner but the effort in these areas was inadequate to say anything substantial. Two of my contributions in PATH were developing the CRiSP passage model and the alpha life cycle model in which the extra mortality was expressed independent of the downstream stocks.

2***. Summarize the major conclusions of your modeling effort relative to the four H's: habitat, harvest, hydropower, and hatcheries. Specifically, state your model's conclusions relative to the following***:

> a. The efficacy of dam breaching or drawdown to natural river levels for delisting ESA species and restoring diverse and productive populations of native fishes throughout the Columbia River Basin?

The weighting of the analyses in PATH indicated that drawdown was better than transportation as a recovery measure for spring chinook while harvest was a possible recovery measure for fall chinook. After the conclusions were developed, publicized and made official, new data from the passage and transportation studies were made available that challenge the PATH conclusions. PATH did not update its analysis to reflect these new findings. The specific PATH conclusions on the probability of recovery with dam breaching vs. transportation are questionable.

> b. The efficacy of hatcheries for delisting ESA species and restoring diverse and productive populations of native fishes throughout the Columbia River Basin? Consider hatcheries that mitigate for lost habitat and those used to supplement depleted stocks.

PATH did not address hatcheries in a scientific manner. Any results were little more than the personal opinions from specific members of PATH and of the review panel.

c. Allocation of harvest and harvest levels needed to delist ESA species and restore diverse and productive populations of native fishes throughout the Columbia River Basin?

The conclusions on harvest levels needed to restore stocks were inextricably bound to the assumptions of the intrinsic productivity of the stocks. The greater the productivity, the more effective harvest as a recovery measure. For both spring and fall chinook, PATH very likely overestimated the intrinsic productivity. This overestimation evolved from the decision support framework that divided PATH into pro and con groups on many issues. Since agendas drove the groups, the hypotheses were formulated with bias towards particular beliefs. Thus, dam removal advocates developed models with high hydrosystem mortality, which required a high intrinsic productivity to balance the large mortality. The group favoring climate as a major contributor to fish decline proposed the hypothesis that the climate would return to pre-1977 conditions, which again inferred a high productivity in the future. The “here to stay” extra mortality hypotheses became so convoluted with the mixture of assumptions intrinsic to PATH’s decision process that it did not accurately represent the possibility that productivity may now be low in the system. The upshot of all this was that the mixture of hypotheses gave very high productivity so harvest reductions predict optimistic recovery probabilities.

d. The efficacy of restoration of tributary and mainstem habitat for delisting ESA species and restoring diverse and productive populations of native fishes throughout the Columbia River Basin?

PATH addressed single species issues and had no essential contribution to restoring diversity in tributaries or the mainstem. Assessments for restoring or creating mainstem fall chinook habitat were approaches in a cursory manner. The discussion of the changes that may occur in the drawn down reservoirs was not linked to the estimated changes in smolt survival or growth.

2. What kinds of information or data are needed to run your model?

The CRiSP passage model has been calibrated with several levels of information. The water quality predictions have been calibrated with physical measurements on total dissolved gas, temperature and flow. Smolt travel time and mortality have been calibrated with the past and recent survival studies using brand release and PIT tag experiments conducted on specific stocks from the Snake and mid-Columbia Rivers. Predictions to evaluate future conditions use a suite of water years representing the expected frequency of low, medium and high water years. Survival for each release is identified in terms of locations and timing of release over the basin. Model predictions on smolt survival require hydrosystem operations, water quality conditions, and stock release information.

3. What are the strengths and weaknesses of your model?

The strength of the CRiSP passage model is that it is calibrated with all the available current data and the model fits the data better than other models developed to describe survival of smolts through the river system. A major weakness of the model is that all fish from a release group are treated as equal. Presently there is no way to characterize differences in accumulated stress from different passage routes or because of pre-smolt conditions. This is particularly significant for subyearling chinook in which size is correlated with survival and migration rate. With the next version of the model, CRiSP 1.7, we are addressing some of these problems. Size and growth are being included through a bioenergetic model. In addition, we will include fish condition through a vitality modeling approach that can deal with the accumulation of stress over time (Anderson in press, Ecological Monographs – "A vitality based model relating stressors and environmental properties to organism survival").

Another issue is the assumption that survival is a function of travel time. The PIT tag studies in the tributaries above the hydrosystem do not support a simple travel time survival relationship as is contained in the CRiSP1.6 model and the FLUSH model. We have developed a new theory to address this problem. It will be included in CRiSP1.7.

4. What are the assumptions of your model?

In the CRiSP 1.6 passage model survival is a function of travel time and environmental water quality including temperature and total dissolved gas levels. In general, survival decreases as an exponential function of travel time. The rate of mortality decrease increases with temperature and when total dissolved gas level is above a threshold level.

The alpha life cycle used in PATH is a variant of the Ricker spawner-recruit model. It assumes climatic variations are characterized by a decadal scale cyclic process and a step process that changes with a cycle of 60 years.

5. How does your model address uncertainty?

Uncertainty in PATH models was expressed through a weighting of the hypotheses by outside experts. In retrospect, this was inadequate and misleading. Since PATH explored a large range of hypotheses, many results could be obtained. Ultimately the weighting by the four review members determined the outcome of the PATH process irrespective of the scientific logic or rigor of the results. Essentially the final PATH results were the reflection of four scientists selected by the moderators. Their written justification for their weighting revealed misconception and biases. The recent survival studies do not comport with their conclusions and the fact that PATH did not reconcile discrepancies between its conclusions and the new data should be of concern to decision-makers. I believe the PATH decisions process did not resolve the uncertainty in the PATH hypotheses. If anything, the degree of certainty in conclusions was overstated.

6. All models make predictions. Why do you think your model's predictions are accurate?

The CRiSP model predictions for spring chinook hydrosystem survival have been proven accurate by comparisons with PIT tag data.

The success in characterizing fall chinook survival is less because the model does not include important behavioral and bioenergetic factors and systemwide estimates of fall chinook survival are currently unavailable.

The life cycle model based predictions on recovery probabilities are likely optimistic.

7. How does your modeling effort relate to or contrast with the other three modeling efforts?

PATH, in spite of problems of balance, openness, and scientific rigor, did explore the bounds of possible outcomes of dam breaching and transportation, and it synthesized much of the data. The other modeling projects have used the PATH data analyses as a foundation for their work. In addition, PATH formalized the mathematical foundations for extra and delayed mortalities.

In our new juvenile passage modeling approach, we are addressing individual smolt characteristics including size, growth, and vitality. This approach is unique in that all other modeling efforts to date have considered fish at a population level that does not differentiate between successful and unsuccessful individuals within the population.

8. What advice would you give decision-makers on how they should use your model to support decisions regarding salmon recovery in the Columbia River Basin?

CRiSP is the best available modeling system to describe the effect of hydrosystem operations on smolt passage survival. It realistically characterizes the effects of flow, spill, gas supersaturation and temperature on fish travel time and survival through the hydrosystem. Neither CRiSP, nor any other passage or life cycle model now available, is adequate to address the issue of delayed mortality or survival of fish above the hydrosystem. To address these issues the models need to be further developed and new experiments need to be planned to characterize the impacts of passage experience on fish survival.

PATH Answers Part III - Experimental Management

From: Charlie Paulsen
Rich Hinrichsen

Date: May 2, 2000

Subj: Reply to ISAB questions for modelers – PATH Experimental Management

Purpose

The complete report (20 pg. Executive summary, 65 pg. Main body, plus appendices) is now on the BPA PATH web page, http://www.efw.bpa.gov/PATH/. We answer the eight questions briefly in this memorandum; more complete information on methods, data, etc. are in the report. The views in this memo are our own (Paulsen and Hinrichsen), although the report proper is a PATH group effort.

In contrast to earlier PATH reports, which focussed on the effects of hydrosystem management actions on Snake chinook population viability, the experimental management (EM) report analyzes trade-offs between biological effects of actions (how they influence persistence and recovery) and how much one could learn by treating actions as experiments whose outcomes are uncertain. It is designed to answer questions like the following:

- If an action increased the ratio of recruits to spawners (at low spawner numbers) by, say, 50% above its recent average, how long would one need to be confident that the action works as planned?
- What are the trade-offs between the length of the experiment and confidence in its efficacy?
- How will population viability be affected by experimenting (e.g., action “on” in odd-numbered years and “off” in even-numbered years) as opposed to turning the action “on” as soon as possible?

Although PATH incorporated many uncertainties via a decision analysis approach, to date no other work within the Columbia has addressed these design questions in detail for ESA-listed stocks.

Major conclusions

In investigating a variety of EM actions, the effects on population viability, and associated learning opportunities, we have looked at a number of life stages, all for Snake spring/summer chinook. These include

- Rearing habitat (increasing egg-smolt or parr-smolt survival via carcass or nutrient enrichment);
- Downstream survival (drawdown of four Lower Snake projects);
- Downstream survival (turn transport on/off in alternate years); and
- Downstream/early ocean survival (reducing Snake hatchery releases).

The model structure (see replies to questions 4, 5, and 6) is quite general, however. The focus is on the recruit/spawner ratio, and the model can accommodate survival changes at any stage in the life cycle. We emphasize how long one would need to monitor population dynamics to detect changes, not on the most probable effects of any given action. Although the effects of the above actions are based on available data, they are uncertain; if one could precisely predict their effects, no management experiments would be required to reduce uncertainty. On one specific point, harvest, spring/summer chinook harvest rules are used by the model to account for changes in harvest rates as a function of changes in the abundance of returning adults. These rates can be changed easily, but past work in PATH suggests that for spring/summer chinook, population viability is not terribly sensitive to changes in harvest rules. We have not performed this type of sensitivity with the EM model.

Model Inputs

The model inputs for "base case" population projections consist of annual estimates of the following for Snake spring/summer chinook. Data series start in 1957, and run through 1994. Note that the model does not rely on upstream/downstream comparisons, or on passage models, for the base case.

- Spawning escapement, estimated from expanded redd counts.
- Spawner age estimates, from scale analyses and length-at-age information. These are often averages from years with available data.
- Estimates of past harvest rates.
- Estimates of upstream survival, from dam counts, "turnoff" into lower river tributaries, and harvest.
- A harvest "schedule" based on abundance in future years and rules for harvest rates as a function of run-year abundance.

For any given action, the model also requires estimates of the effects on life-cycle survival. The data sources for these are documented in the body of the EM report. Examples include correlations between spawning escapement and parr -> smolt survival estimates (latter from Paulsen and Fisher, 2000, in review), or correlations between recruit/spawner ratios and hatchery releases. For drawdown actions **only**, we use passage model output to estimate effects of drawdown on life-cycle survival.

Information Needs

Beyond the data noted above, the model needs a schedule for when EM actions will be implemented (e.g., on/off in odd/even years for 10 years running). Similar information on a stock-specific basis is required for actions that may effect only a subset of the index stocks (e.g., carcass/nutrient enrichment).

Strengths and weaknesses

There are two important strengths of the model. First, as a population projection model, it is very straight-forward regarding data requirements and the basic mechanics of the population dynamics:

$$\ln(R_{i,t}/S_{i,t}) = a_i + b_i S_{i,t} + m_t + \Delta m_{i,t} + e_{i,t}$$
$$R_{i,t,age} = mature_{i,t,age} \times R_{i,t}$$
$$S_{i,t} = prespwn_{i,t} \times conv_{i,t} \times (1 - mharv_{i,t}) \times (1 - tharv_{i,t}) \times \sum_{age} R_{i,t-age,age}$$

i = subscript representing index stock
t = subscript representing brood year
$R_{i,t}$ = recruits (adult offspring) from parent spawners $S_{i,t}$
$S_{i,t}$ = spawners returning in year t
a_i = average Ricker-a
b_i = Ricker-b
m_t = common deviation from average productivity
$Dm_{i,t}$ = hypothetical changes in productivity schedule (E.M. Design)
$e_{i,t}$ = Normally distributed process + counting error (white noise)
$R_{i,t,age}$ = age-specific recruits (adult progeny)
$mature_{i,t-age,age}$ = age-specific maturity schedule
$prespwn_{i,t}$ = pre-spawning survival
$conv_{i,t}$ = upstream survival (conversion rate)
$mharv_{i,t}$ =mainstem harvest rate
$tharv_{i,t}$ = tributary harvest rate

Estimation of the model is accomplished by doing a least squares fit of
$\ln(R_{i,t}/S_{i,t}) = a_i + b_i S_{i,t} + m_t + e_{i,t}$
to the spawner-recruit data. The model contains 264 observations and 52 parameters (including sigma^2).

The only complications (in population projection) arise because of the necessity of tracking multiple age-structured populations through time.

Second, it seem to us that it is well-suited for estimating both the biological (population dynamics) and information (experimental power) effects of EM management regimes. These can be as simple or complex as desired by the model user, effecting all stocks permanently (e.g., drawdown) or different stocks in different ways over time (e.g., different nutrient enrichment schedules for different stocks and years). In the report, we show examples of the model's use with a variety of ancillary data to estimate the effects of EM actions, ranging from PIT-tag based SAR's to hatchery releases and parr -> smolt survivals derived for studies designed to detect arrival times at Lower Granite.

The model's main weaknesses are the strong reliance on spawner-recruit estimates, which may contain systematic errors (random variation should be absorbed into the process error terms in the model), and the basic assumption that the estimates of an action's effects (which are model inputs) are accurate. On the other hand, if the effects of any given action were known with certainty, there would be no need for an experiment, and little need for population dynamics models.

Assumptions

Many assumptions are required for this or any population dynamics model. Among the most important are:

- Future conditions will be like 1978-1994, absent management intervention;
- Harvest rules, conversion rates, and population age structure will be similar to recent respective means and variances;
- Joint distributions of estimated model parameters (Ricker a_i's and b_i's, m_t's) apply to the future.

For detecting the effects of EM actions, the assumption regarding future conditions like 1978-1994 can be relaxed, so long as changes in other conditions (e.g., climate) do not coincide with the schedule for the EM action (e.g., on in odd years, off in even years).

Uncertainty

The uncertainty in parameter estimates is assessed by summarizing the sampling distribution via confidence intervals. The population projection model respects parameter uncertainty by drawing from the joint sampling distribution of the estimates using technique of Gelman *et al*. (1995, p. 237) which treats the joint sampling distribution as a "posterior" distribution or using "bootstrapping the residuals" (Efron and Tibshirani 1993, p.113-115). Uncertainty in effects of management actions is addressed using (optional) distributions of projected effects. Uncertainty in risk measures is assessed using confidence intervals.

We assume that statistical distributions estimated from historical data will continue, and that EM actions are not perfectly confounded with other actions or natural phenomena. The latter is of most concern with actions that can be turned off or on only once (e.g., drawdown).

Prediction accuracy

We have tested the fit of the retrospective model extensively, including searches for outlying data points, addition of more index stocks (another 5-10 have been developed), excluding/including recent years of spawner/recruit data (1991-1994), and a variety of other statistical techniques, and found it to be robust.

Of course, the future is unknown. If, for example, the 2000 returns of spring chinook, currently running 3-5 times the 10-year average, signal a long-term change in marine survival, the model's assumption about low-survival recent conditions will not hold. On the other hand, the EM design should be robust to most such changes.

Other modeling efforts

This is the only modeling effort we are aware of that has done a systematic exploration of trade-offs among population risk measures, experimental design, and knowledge gained via an experimental approach to reducing uncertainty. Initial steps by CRI in this regard (McClure *et al.* 2000, section V-E) basically consist of before-and-after comparisons of abundance for isolated populations, which can of course be influenced by a myriad of natural and anthropogenic factors. The parr -> smolt survival estimates used as one approach for estimating the effects of nutrient additions are drawn from one of very few studies that directly estimate freshwater survival rates as a function of land use patterns.

Table. Current differences in CRI and EM modeling at a glance.

	CRI (Dennis-type models) McClure *et al*. (2000)	EM (Screening) Model
Covariability in growth rates among stocks	No (in general), but index stocks combined into single population for some analyses.	Yes
Can project responses to experimental treatments interspersed over time and space.	No	Yes
Recovery escapement thresholds	No	Yes
"Survival" escapement thresholds	No	Yes

Table. Current differences in CRI and EM modeling at a glance.		
	CRI (Dennis-type models) McClure *et al*. (2000)	EM (Screening) Model
Extinction probabilities	Yes	Yes
Confidence intervals reported for:	lambda (for ESU-level only)	All model parameters, probability of extinction and exceeding recovery escapement, fraction of years survival escapement exceeded.
Density dependence	No	Yes. Growth rate decreases with increased population size.
SR “data” time span	BY 1980-1994	BY 1957-1994
Growth rate distribution	Instantaneous growth rate is Gaussian white noise (no trend)	Density-independent growth is decomposed into common growth effect (allows for trend) + white noise
Population response	ln(N(t+1)/N(t)), where N(t) is a “weighted running sum”	ln(Recruits/Spawner), where recruits are adult progeny arriving at Bonneville Dam.
Time step	Continuous. Reproduction modeled as continuous process	Discrete. Reproduction is seasonal.
Observation error in SR data	Handled using innovative estimation technique and “weighted running sum”	Limited weir counts used to estimate ratio of process to observation error variance.
Achieving target viability	Uses changes in lambda	Uses changes in log(Recruits/Spawner), upstream survival, harvest

Advice

Our advice would be to take the population dynamics predictions with several grains of salt: changes in climate and other uncontrollable factors make predictions of extinction times, probabilities, etc. very uncertain. However, we have more confidence in the overall results from the EM modeling: actions that affect all stocks, even if tested rigorously over time periods of one or more decades, will likely have substantial residual uncertainty in their effects. These "temporal" controls (on/off) must run for many years because of high year-to-year variation in survival. Experiments that utilize more traditional "spatial" controls (e.g., treat half the index stocks, using the remainder as controls) have far more power to reduce uncertainty.

References:

Efron, G. and R.J. Tibshirani. 1993. An introduction to the bootstrap. Monographs on Statistics and Applied Probability 57. Chapman & Hall/CRS, New York.

Gelman, A., J.B. Carlin, H.S. Stern, and D.B. Rubin. 1995. Bayesian data analysis. Chapman & Hall. New York.

McClure *et al*. 2000. Draft document. Standardized quantitative analysis of the risks faced by salmonids in the Columbia River Basin. Available at the CRI web site.

Ecosystem Diagnosis and Treatment Answers

ISAB Model Review

Response to Questions Relative to the Ecosystem Diagnosis and Treatment Method.

1. ***What is the purpose of your modeling effort? What questions or problems were your models designed to address?***

The EDT method is a landscape-based approach for relating events and actions affecting animal populations and their habitat to the long-term performance of species of interest (Lichatowich et al. 1995; Mobrand Biometrics, Inc. 1999). The EDT method incorporates a conceptual framework, a procedure, and a set of tools (Lestelle et al. 1996). The framework provides the theoretical foundation, the procedure prescribes the steps in planning and analyzing ecosystem information in support of decision-making, and the tools are the databases and model components used to organize, analyze, and summarize information.

Development of EDT was funded largely through the NPPC Fish and Wildlife Program and in response to "concerns raised by the ISAB and its predecessors (ISG and SRG, e.g. ISG 2000) regarding the need for a clear and explicitly stated framework to serve as the rational, scientific basis for the FWP" (NPPC 1997; Mobrand et al. 1998).

EDT was designed to address cumulative effects of many environmental factors, in many places. It is a tool for tracking the operating assumptions upon which adaptive management action plans can be built. EDT can be used to:

- develop working hypotheses regarding action priorities; for example, diagnose salmon performance problems and identify treatment alternatives;
- develop and document working hypotheses—the conditions required to achieve specified objectives—for detailed and comprehensive action alternatives;
- assess potential benefits and risks associated with incorrect operating assumptions—in other words, analyze consequences under alternate hypotheses;
- identify key uncertainties to guide research and monitoring and evaluation.

EDT was designed to be an expert system and, as such, differs from statistical models in both purpose and design (Holling 1998, Hilborn and Mangel, 1997)—a key distinction. Expert systems seek to be comprehensive and explanatory, they generate hypotheses rather than test them, and they formulate operating assumptions upon which actions can be based and risks understood. C.S. Holling (1999) states the distinction in another way "*…reductionist science, i.e. the science of parts, was essential to provide bricks for an edifice, but not the strategic design of the edifice. Such strategic design is needed for appropriate diagnosis and policy, and it has to emerge from a science of integration. A* *science of integration combines research and application, is interdisciplinary and faces the realization that knowledge of the system we deal with is always incomplete."*

EDT is a tool for building hypotheses. It depends upon the results of statistical analyses to formulate new hypotheses, which in turn can be tested through statistical models. It is a framework for integrating results from statistical analyses with knowledge and opinions from all relevant scientific disciplines.

Expert systems like EDT consist of data linked to conclusions through a set of rules. EDT was conceived to be flexible so that new knowledge, information, and data can be incorporated, thus allowing the expert system to remain current. At the heart of EDT is the structure that defines how data and rules are linked. This temporal-spatial-conceptual architecture of data and rules is the Framework as defined and explained in the Multi-species Framework process (NPPC 1997; EWG 1999; Mobrand et al. 1997; Lestelle et al.1996). The data and rules themselves are more ephemeral and replaceable.

EDT computes future capacity, productivity and diversity of salmon in the Columbia Basin as a function of: a) the bank of genetic resources embodied by the animals alive today and b) the environment available to them. The current version of the EDT expert system, while incorporating genetic fitness factors, has a primarily environmental focus. EDT constructs "survival landscapes" that describe the patterns of survival conditions for salmon over time and space. EDT assesses the *quality* of the habitat in regard to the biological template of specific species, such as chinook salmon (Mobrand et al. 1997).

The specific purpose of the application of EDT in the Multi-species Framework analysis was to compare a set of comprehensive visions for the Columbia Basin in terms of the prospective performance of salmon populations in each of ten provinces (McConnaha 1999). The first stage in this analysis focused on chinook salmon. Based on available environmental descriptions of some 7,500 spatial habitat units (6-level HUCs) for the Columbia River basin as well as information on hatchery production, harvest, and hydro operations, we projected productivity, capacity and diversity for 73 natural and 50 hatchery populations of chinook salmon under 10 different scenarios. These scenarios included the current, historic, no action conditions as well as seven alternative futures for the Columbia River Basin. The analysis represents the long-term average performance potential for chinook salmon.

2. ***Summarize the major conclusions of your modeling effort relative to the four H's: habitat, harvest, hydropower, and hatcheries. Specifically, state your model's conclusions relative to the following:***

The EDT analysis performed so far for the Columbia Basin estimates cumulative effects of all H's. We have not attempted to analyze isolated strategies; instead, the Multi-species Framework analysis compared alternatives composed of prescribed mixes of all H's. Each of these alternatives was based upon a *worldview* and was modeled using [operating] assumptions consistent with this worldview (Costanza 2000). As we finalize the Multi-species Framework analysis, we will address the biological consequences when alternate worldviews are true, in an attempt to shed some light on the benefits and risks associated with strategies that emphasize some of the H's more than the others.

Because we have not completed our strategies analysis, the answers below represent general impressions of results so far.

a. The efficacy of dam breaching or drawdown to natural river levels for delisting ESA species and restoring diverse and productive populations of native fishes throughout the Columbia River Basin?

EDT and the Framework Project did not attempt to address the needs for delisting ESA species. Drawdown and dam breaching were effective strategies for restoring mainstem chinook habitat with a relatively low level of biological risk but with a great risk of social change and political will.

b. The efficacy of hatcheries for delisting ESA species and restoring diverse and productive populations of native fishes throughout the Columbia River Basin? Consider hatcheries that mitigate for lost habitat and those used to supplement depleted stocks.

The efficacy of hatcheries to replace natural habitats and ecological functions is not an analytical problem but will only be solved by careful study. Absent firm information, our approach has been to bracket alternatives that make heavy use of hatcheries by using a range of effectiveness assumptions. Our initial assumption was that hatcheries fulfilled their promise and were comparably effective as natural populations. We are contrasting this with a view that says they are substantially less effective than natural populations.

c. Allocation of harvest and harvest levels needed to delist ESA species and restore diverse and productive populations of native fishes throughout the Columbia Rover Basin?

The alternatives that we examined assumed different harvest allocations and rates, including strategies that greatly reduce harvest rates. In general, harvest reductions had little impact on spring chinook stocks but were important to the abundance of fall chinook. We have not attempted to assess impacts of harvest rates on ESA status.

d. The efficacy of restoration of tributary and mainstem habitat for delisting ESA species and restoring diverse and productive populations of native fishes throughout the Columbia River Basin?

We looked at a wide array of strategies to change habitat attributes in tributaries and mainstem areas. Breaching dams, for example, was quite effective in restoring mainstem habitats to the benefit of chinook performance. Land-use changes and practices in tributaries were, over the long term, effective in restoring abundance and distribution of tributary populations.

3. What kinds of information or data are needed to run your model?

EDT requires qualitative and quantitative habitat information (baseline input data) and generic information about the salmon species and their habitat requirements (rules relating survival to habitat conditions). Habitat information is collected at the HUC-6 level (about 7,500 "pixels" in the U.S. portion of the Columbia River basin) in regard to some 40 habitat parameters. Biological information is collected at the population level. For chinook salmon, we have described 73 natural populations in the basin.

4. What are the strengths and weaknesses of your model?

Perhaps the greatest strength of the EDT approach is that it accounts for cumulative effects—such as spatial temporal interactions, all attributes, competition, predation effects. In addition, EDT:

- complements and is supported by statistical approaches,
- enhances the value of research and monitoring results,
- computes population performance in terms of productivity, capacity, and diversity,
- translates combinations of actions at any scale into biological performance responses,
- examines comprehensive alternative futures,
- diagnoses problems and prioritizes treatments,
- provides a repository of knowledge and the means for documentation and accountability,
- provides a conceptual framework for understanding,
- identifies critical uncertainties (for directing monitoring activities and research),
- separates policy from science,
- describes benefits and risks of approaches,
- supports trade-off analysis.

Among the weaknesses of the current version of the EDT model are:

- lack of tools for easy review and edit of all rules, (we are working to remedy this problem)
- need for ground truthing of input data and routine peer review process to ensure that the rules are well understood and consistent with current information and knowledge, (a structured, independent certification process—with standards for documentation and access—for data and rules would greatly enhance the utility of this and other models)
- need for genetics components to be expanded to accommodate a wider range of assumptions
- because of the dimensionality (many reaches, time periods, life stages and attributes) the model is demanding in terms of computer resources (it takes hours to run and requires lots of memory).

5. *What are the assumptions of your model?*

The fundamental assumption of EDT is that the Multi-species conceptual foundation and framework are appropriate and useful to the decision-making process they are intended to support. Specifically, EDT assumes that habitat is the template for biological performance (Southwood 1977). EDT begins with a detailed habitat description and links it to ecological function and biological performance of specific species.

For the most part, EDT is a flexible, expert system capable of incorporating a wide range of hypotheses and assumptions. Each scenario modeled represents a unique set of assumptions (worldview) and a specified suite of actions.

Two parameter density-dependent survival functions are assumed (Mousalli and Hilborn 1986) for each life cycle segment. Typically the Beverton-Holt equation is used; others can be accommodated.

6. *How does your model address uncertainty?*

EDT is a steady-state model that predicts long-term outcomes. It does not incorporate stochastic elements. It includes current population status only as genetic fitness factors and, therefore, does not predict the length of time required to reach the long-term state. It does not explicitly compute the extinction risk.

The implications of uncertainty are addressed through sensitivity analysis, which in turn can guide recovery actions as well as monitoring and research efforts towards risk averse solutions.

7. *All models make predictions. Why do you think your model's predictions are accurate?*

"All models are wrong. Some are useful." (a paraphrase of a statement for which I can't recall the source). EDT provides an accurate depiction of the environment and biological performance given our current scientific understanding and available data. Its depiction, like that of any model, is imperfect because our knowledge is imperfect and because nature is inherently complex. It does however provide a framework within which we can accumulate knowledge and improve predictability over time.

EDT inputs consist of detailed descriptions of the landscape. It captures the obvious features of the landscape on a fine scale in terms of many observable attributes. For example, it includes length and width information of stream reaches; it routes these reaches thus accounting for the connectivity (or lack thereof) along the salmon migration pathways. The unique conditions of a watershed described this way, coupled with the relatively well understood habitat requirements and patterns of movement of salmon, allow EDT to predict steady-state population parameters and to compare relative effects of different habitat changes on salmon performance in a way that is useful and informative to decision making.

In other words, the main reason EDT provides *useful predictions* is that it captures the habitat conditions on a fine scale and analyzes their significance from the point of view of salmon and other species.

8. ***How does your modeling effort relate to or contrast with the other three modeling efforts?***

The relationship between CRI, PATH, and EDT is complementary. The EDT analysis can and should incorporate the analytical results of the statistics-based PATH and CRI. The EDT model can be used to develop hypotheses regarding, for example, the significance of specific habitat survival relationships, which in turn might be tested with statistical models. CRI fits parameters to observed population data. The EDT model is driven by habitat data and does not input any population numbers. See Kareiva et al. 1999.

The ICBEMP method is similar to EDT in concept and data requirements. EDT estimates quantitative and cumulative effects of actions on salmon populations.

The CRITC model offers some very intuitive ways for the user to explore alternative strategies and assumptions. EDT might complement this model by expanding its analytical capabilities. Our impression is that the strength of the CRITFC model relative to EDT is in the user interface; whereas EDT is analytically more powerful.

9. ***What advice would you give to decision-makers on how they should use your model to support decisions regarding salmon recovery in the Columbia River Basin?***

We believe that models can best aid decision-makers by highlighting risks and tradeoffs of alternative actions. No model *does it all* for most decisions. For this reason, we strongly favor the concept of an analytical toolbox. EDT can and should be used in conjunction with other models such as the CRI, PATH, and CRITFC tools. This would allow the region to take advantage of the strengths of each model system and to appreciate the different perspectives afforded by each. Unfortunately, models too often become analytical bludgeons in political warfare. They are used to bolster political positions based more on value sets than on scientifically derived information. We find that this is the cause of much of the mistrust of models by decision-makers.

One approach that we are using to inform decision-makers is to analyze alternatives in light of different worldviews and assumptions (Costanza 2000). This is an attempt to highlight the underlying tradeoffs of different alternatives and their vulnerability to both social risk (will we *really* make the social changes necessary?) and biological/ technological risk (can technology effectively substitute for natural ecological function?). We believe that this type of analysis would be complemented by analysis of extinction risk for specific populations, for example, that could be provided by other models such as CRI.

Hypothetical Decision Matrix		**Strategies/Actions**		
Worldview	**Key Assumptions (4H's)**	**Regulation Restoration …**	**Mix**	**Hatcheries Transport …**
Holistic-Multispecies		**Vision A**	Vision B	Vision C
Intermediate		Vision D	**Vision E**	Vision F
Agricultural		Vision G	Vision H	**Vision I**

EDT would flesh out the Visions in this decision chart. Operating assumptions apply on the diagonal, alternate assumptions off the diagonal.

Reference Cited

Costanza, R. 2000. Visions of alternative (upredictable) futures and their use in policy analysis. Conservation Ecology [online] 4(1): URL: http://www.consecol.org/vol4/iss1/art5

Hilborn, R. and M. Mangel. 1997. The Ecological Detective: Confronting models with data. Princeton University Press. Princeton, NJ.

Holling, C. S. 1999. Introduction to the special feature: just complex enough for understanding; just simple enough for communication. *Conservation Ecology* **3**(2): 1. [online] URL: http://www.consecol.org/vol3/iss2/art1

Holling, C.S. 1998. Two cultures of ecology. Conservation Ecology [online] **2**(2): 4. Available from the Internet. URL: http://www.consecol.org/vol2/iss2/art4

Independent Scientific Group 2000. Return to the River 2000: restoration of salmonid fishes in the Columbia River Ecosystem. Portland, OR, Northwest Power Planning Council.

Lestelle, L.C. 1996, L.E. Mobrand, J.A. Lichatowich, T.S. Vogel. Applied Ecosystem Analysis – A Primer: The Ecosystem Diagnosis and Treatment Method. Bonneville Power Administration. Portland, OR.

Lichatowich, J. A., L. E. Mobrand, et al. 1995. An approach to the diagnosis and treatment of depleted Pacific salmon populations in freshwater ecosystems. Fisheries 20(1): 10-18.

McConnaha, W. E. 1999. The Columbia River Multi-Species Framework Project: Progress and Concepts. Seattle, WA, American Water Resources Association. Proceedings, December 1999.

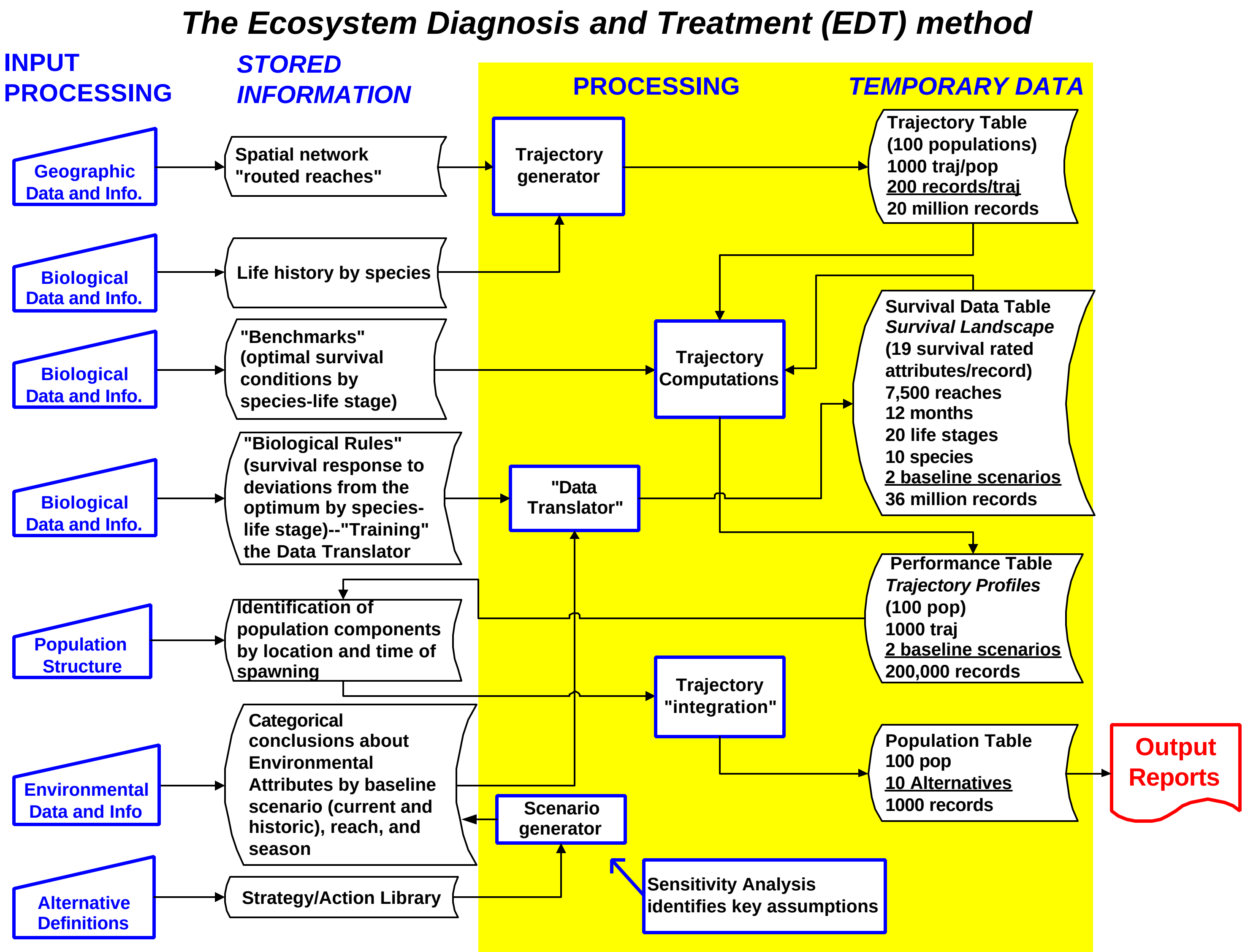
Mobrand, L.E., L.C. Lestelle, G.R. Blair, 1998. Recovery of a Columbia River Watershed from an Ecosystem Perspective. A case study using the EDT method. Final Report. Bonneville Power Administration. Portland, OR.

Mobrand, L. E., J. A. Lichatowich, et al.1997. An approach to describing ecosystem performance "through the eyes of salmon". Canadian Journal of Fisheries and Aquatic Sciences 54: 2964-2973.

Mobrand Biometrics, Inc. 1999. The EDT Method. Portland, OR, Northwest Power Planning Council.

Moussalli, E. and R. Hilborn 1986. Optimal stock size and harvest rate in multistage life history models. Canadian Journal of Fisheries and Aquatic Sciences 43(1): 135-141.

Northwest Power Planning Council 1997. An integrated framework for fish and wildlife managment in the Columbia River Basin. Portland, OR, Northwest Power Planning Council.



The Ecosystem Diagnosis and Treatment (EDT) method
INPUT PROCESSING
STORED INFORMATION
PROCESSING
TEMPORARY DATA
Geographic Data and Info.
Spatial network "routed reaches"
Trajectory generator
Trajectory Table (100 populations) 1000 traj/pop 200 records/traj 20 million records
Biological Data and Info.
Life history by species
Biological Data and Info.
"Benchmarks" (optimal survival conditions by species-life stage)
Trajectory Computations
Survival Data Table Survival Landscape (19 survival rated attributes/record) 7,500 reaches 12 months 20 life stages 10 species 2 baseline scenarios 36 million records
Biological Data and Info.
"Biological Rules" (survival response to deviations from the optimum by species-life stage)--"Training" the Data Translator
"Data Translator"
Performance Table Trajectory Profiles (100 pop) 1000 traj 2 baseline scenarios 200,000 records
Population Structure
Identification of population components by location and time of spawning
Trajectory "integration"
Environmental Data and Info
Categorical conclusions about Environmental Attributes by baseline scenario (current and historic), reach, and season
Population Table 100 pop 10 Alternatives 1000 records
Output Reports
Scenario generator
Alternative Definitions
Strategy/Action Library
Sensitivity Analysis identifies key assumptions

CRITFC Cohort Model Answers

Submitted by: Jeff Fryer, Phil Roger

1.) What is the purpose of your modeling effort? What questions or problems were your models designed to address?

a. A communication and education tool

The cohort model has an easy-to-use interface designed to allow users to test salmon recovery options. The user can easily manipulate harvest rates, predator control efforts, smolt transportation rates and assumptions, habitat restoration efforts, as well as environmental variables. The model allows users with all levels of knowledge of salmon issues to design salmon recovery scenarios and determine their result. Graphical output is provided giving the resulting population growth rates under baseline conditions, user-modified conditions, and user-modified conditions with optimum habitat, hydro, and no harvest scenarios.

b. Stimulate users to provide additional information if they disagree with part of the model

The data and equations used by the model are available to the user. If the user disagrees with the either the data or equations, the user is welcome to supply their own, which could then be incorporated into the model.

c. Demystify models and provide wider user access to quantitative tools

By providing an easy-to-use interface, making the data and equations readily available to the user, and also providing references for data values, we have provided a model that we hope will be widely used and inspire some level of confidence in its results among users.

d. Provide the essence of several complex tools in a single package

1. PSC/PFMC ocean harvest analyses
2. TAC in-river harvest
3. PATH downstream passage
4. EDT subbasin habitat

e. Allow users to do their own trade-off analysis between the four “H’s”

The user can manipulate parameters affecting each of the “4 H’s”. Population growth rates are calculated successively for the user-selected scenario with optimum subbasin habitat, no harvest, and under an “optimal hydro” scenario consisting of the breaching of the lower Snake River dams plus a drawdown of John Day reservoir. This provides some measure of the impact of three of the four “H’s” on the scenario selected by the user.

2.) Summarize the major conclusions of your modeling effort relative to the four H's: habitat, harvest, hydropower, and hatcheries. Specifically, state your model's conclusions relative to the following:

OUR MODEL IS STILL UNDERGOING DEVELOPMENT AND TESTING. WE HAVE NOT REACHED FINAL CONCLUSIONS ON THE QUESTIONS POSED. THE FOLLOWING STATEMENTS ARE PRELIMINARY AND SUBJECT TO CHANGE.

a. The efficacy of dam breaching or drawdown to natural river levels for delisting ESA species and restoring diverse and productive populations of native fishes throughout the Columbia River Basin

We have modelled only an aggregate of the Middle Fork Salmon R. spring chinook populations in the Snake R. To the extent this aggregate is representative of all Snake River spring chinook populations, we draw the following conclusions:

1. Spring chinook naturally reproducing population growth rate is below replacement and cannot be rebuilt without dam breaching.

2. Even with a high rate of supplementation, it is difficult to achieve a population growth rate above replacement without survival gains from the other H's.

3. Survival gains from other H's are needed to allow the population to rebuild, even with dam breaching

4. Breaching 4 Snake River dams does nothing to help populations outside the Snake River (except possibly by improving flows, which we have not modelled).

5. Drawing down John Day to spillway crest produces some benefits for all stocks originating above that point.

b. The efficacy of hatcheries for delisting ESA species and restoring diverse and productive populations of native fishes throughout the Columbia River Basin? Consider hatcheries that mitigate for lost habitat and those used to supplement delisted stocks.

We have considered only the demographic effects of hatcheries at this time. From this perspective, hatcheries are a very effective tool for maintaining and rebuilding existing populations and as a source for reintroducing salmon into areas from which they have been extirpated.

c. Allocation of harvest and harvest levels needed to delist ESA species and restore diverse and productive populations of native fishes throughout the Columbia River Basin?

Even total elimination of all harvest, by itself, is not sufficient to maintain or rebuild spring chinook populations in the Snake River or above Priest Rapids dam.

d. *The efficacy of habitat improvements for delisting ESA species and restoring diverse and productive populations of native fishes throughout the Columbia River Basin?*

Other than EDT, this is the only model that attempts to represent the effects of improving habitat on population growth.

e. Other conclusions?

The relative proportions of the H's needed to recover spring chinook populations varies in important ways in different areas of the Columbia Basin. For example, breaching Snake River dams benefits Snake River populations, but is ineffective for recovering other populations. Habitat improvements become more effective as one moves downriver and the effects of the hydrosystem decreases. Maintaining some populations appears impossible without using artificial production technology.

3.) What kinds of information or data are needed to run your model?

The data required to run the model and the associated documentation, and the equations used, are all accessible through the model. The Cohort Model is an age based predictive model based on the work of Thompson and Bell (1934) as described in Sanders (1995) that proceeds successively from youngest to oldest age classes. Salmon life history is broken into a series of stages with the number of fish leaving one stage becoming the input for the next stage. Instantaneous mortality rates are calculated for the various mortality factors for each life stage and then applied to determine the number of fish entering the next stage. The data needed are those necessary to calculate the instantaneous mortality rate at each life stage. This data includes subbasin habitat data such as temperature and sediment data (or stream condition data which is transformed into these values), mortality rates at various life history stages (such as overwintering, estuary/early ocean, and annual ocean mortality rates), as well as harvest rates.

4.) What are the strengths and weaknesses of your model?

Strengths:

a. Runs very quickly on a mid-range PC
b. Easy for new users to use
   The model has a simple user interface allowing anyone to run the model. We are working on making the model available via the Internet.
c. Structure, data sources, and assumptions are very transparent
   All are easily accessible to the user running the model. Data can be easily modified by making changes to a Microsoft Access database.
d. Mimics the general behavior of several more detailed models in a single package
e. One of the few models to include the potential effects of habitat change on population growth

Weaknesses:

a. There are no density dependent effects.
b. There is no stochasisity
c. All modifications are instantaneous-there are no time lags built in for hydrosystem or habitat changes.
d. Population sizes are not projected into the future.
e. The effects of changes in flow are not modeled.

5.) What are the assumptions of your model?

Major assumptions:

- Spawning and rearing are evenly distributed throughout currently used habitat regardless of habitat quality.
- Habitat improvements are applied first to the lowest quality habitat.
- Habitat can only be restored to "good" condition.
- A per-dam mortality rate of 17%.
- John Day drawdown reduces mortality by ½ of a dam.
- Hatchery fish upon release are treated as being identical to natural origin fish with the exception of a 50% post release mortality applied to the hatchery fish

6.) How does your model address uncertainty?

Uncertainty is not addressed except to allow the user to specify what the user thinks are the correct values. The user can therefore conduct a sensitivity analysis on variables that are of concern.

7.) All models make predictions. Why do you think your model's predictions are accurate?

Population growth rates from the model for spring chinook are of similar magnitude to observed rates estimated independently by others (Petrosky, C.E. et al. 1995).

8.) How does your modeling effort relate to or contrast with the other three modeling efforts?

- We have attempted to represent the general behavior of PATH, PSC, and TAC analyses without much of the complexity and detail.
- We intend to compare the subbasin behavior of our model with EDT. The original intention was to see if we could use EDT smolt output as the input to our model. If successful, one might then call this model "EDT lite".
- We have not compared our efforts to the CRI model. We are beginning this effort with local NMFS staff. Analytically, there seems to be much similarity between our methods.

9.) What advice would you give decision-makers on how they should use your model to support decisions regarding salmon recovery in the Columbia River Basin?

- This model can be an effective tool for teaching local watershed councils about the impact of conditions outside individual subbasins upon local restoration efforts.
- The model can be used for broad scale trade-off analyses across the 4-H's. Additional detail can be obtained by using more detailed models, if needed. For example, our model could be used to approximate the amount of benefit possible from habitat improvements, followed by a local EDT analysis to determine which sets of actions would be most effective within a subbasin.

Literature Cited:

Petrosky, C.E., H.A. Schaller, and R.C. Beamesderfer. 1995. Spawner - recruit relationships for spring and summer chinook salmon populations in several Columbia and Snake River subbasins. Unpublished.

Sanders, M.J. 1995. Introduction to Thompson and Bell yield analysis using Excel spreadsheets. FAO Fisheries Circular 895. Rome.

Thompson, W.F. and F.H. Bell. 1934. Biological statistics of the Pacific halibut fishery. 2. Effect of changes in intensity upon total yield and yield per unit of gear. Rep. International Fishery (Pacific Halibut) Commission.

\\nwppcu1\fish\em\ww\isab\1 isab final reports\isab 2001-1 model synth.doc