ANALYTICAL APPROACHES TO ASSESSING RECOVERY OPTIONS FOR SNAKE RIVER CHINOOK SALMON.

A scientific summary prepared

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Vancouver, WA

Citation: Budy, P. Analytical approaches to assessing recovery options for Snake River chinook salmon. UTCFWRU 2001(1): 1-86.

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EXECUTIVE SUMMARY

The depressed status of Snake River stocks of chinook and steelhead and the recent listings of many salmon stocks in the Columbia Basin have led to several analytical evaluations and management advice aimed at recovery of these stocks. These different analytical reviews address the effectiveness of different hydrosystem options as well as the potential for recovery through improvements that increase survival at other life stages (e.g., habitat, harvest). Hydrosystem options evaluated included status quo, maximizing transportation, and the option of breaching the lower four dams on the Snake River (also called drawdown and natural river options), the main topic of the Lower Snake River Juvenile Salmon Migration Feasibility Report / Environmental Impact Statement (USACE).

The first review was completed by PATH (Plan for Testing and Analyzing Hypotheses), an open forum composed of modelers, fishery biologists and statisticians from all three states (Oregon, Washington, and Idaho), the federal government (Army Corps of Engineers (USACE), US Fish and Wildlife Service (USFWS), National Marine Fishery Service (NMFS), Bonneville Power Administration (BPA), the treaty tribes of the Columbia Basin (represented by the Columbia River Inter-tribal Fish Commission -CRITFC), and the Northwest Power Planning Council (NPPC). The PATH approach was based on a decision analysis that showed which management actions are the most robust to remaining uncertainties (i.e. the least risky) and allows a decision to be made with full consideration of uncertainty and risk. PATH analyses were followed by the NMFS effort called CRI- the Cumulative Risk Initiative. CRI analyses explore the demographic effects of hypothetical reductions in mortality at different life stages based on current conditions. PATH and CRI analyses were followed by an analytical comparison of their approaches and results completed by a sub group of PATH composed of scientists from the states of Oregon, Idaho, Washington, CRITFC, and the USFWS. In addition, specific analyses have considered the potential for improvement at certain life stages (e.g., freshwater spawning and rearing; Petrosky et al., *in press*) and key uncertainties that affect the likely effectiveness of dam breach (e.g., delayed hydrosystem mortality; Budy et al., *in review*).

This annex synthesizes analyses and results PATH, NMFS CRI, and comparative and follow-up analyses which have been completed since and are summarized here and described in greater detail elsewhere (see below).

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PATH

Although the results vary some among approaches, all available science appears to suggest that dam breach has the greatest biological potential for recovering Snake River salmon and steelhead. Prospective PATH analyses, or consideration of possible <u>future</u> management actions indicated that the dam breach actions had higher probabilities of achieving the survival and recovery standards than the "transportation" actions. Dam breach actions met the standards over a wide range of assumptions (i.e. these actions are robust to remaining uncertainties). The dam breach actions were also <u>less risky</u> than the "transportation" actions (i.e. model projections had relatively little variability over the full range of assumptions).

PATH also performed detailed sensitivity analyses to a range of habitat improvements, additional predation (e.g., avian), and harvest restrictions. These analyses demonstrated that while it is likely that habitat improvements for some spring and summer chinook and steelhead stocks will increase the productive potential in the Snake River Basin, it <u>does not</u> appear that <u>habitat improvement alone</u> would be enough to recover these stocks. Additional predation did <u>not</u> change the rank of management actions, and given the existing low levels of harvest on spring/summer chinook it is unlikely that additional future reductions in harvest alone will significantly increase the survival of spring/summer chinook such that jeopardy standards will be meet. Harvest reductions were sufficient to reduce the threat of extinction for fall chinook only under the most dramatic scenario. However, it is important to note that these extremely dramatic changes would have to be in place forever. As soon as harvest restrictions were relaxed, the fall population would likely drop below recovery levels again.

NMFS CRI

For prospective analyses, NMFS evaluated the effect of eliminating all direct migration mortality except a small tribal subsistence harvest. This direct mortality includes only mortality that occurs immediately in the hydrosystem from turbines, bypass, and ladders etc... but does not include delayed or latent mortality that may be caused by the stress of hydrosystem passage or transportation but occurs later, in the estuary and ocean. This analysis showed that even with 0% mortality in river for juveniles and adults, stocks would continue to decline (Karieva et al., 2000). From these types of scenarios NMFS concluded that by itself, dam breaching is unlikely to recover spring/summer chinook salmon.

In addition, NMFS evaluated the effects of improvements in survival at other life stages (other than in-river migration) and whether these improvements would be enough to halt the decline in the absence of dam breach. They first concluded in the draft A-Fish Appendix (NMFS 1999) that recovery of Snake River spring/summer chinook is likely to require dam breaching and many other aggressive management actions. Later analyses considered combinations of egg-to-smolt survival and early estuary and ocean survival that would halt the decline in the population (Karieva et al., 2000). These analyses showed that egg-to-smolt mortality must be reduced by 11% or early ocean /estuarine mortality by 9% to reduce the risk of extinction to acceptable levels. Delayed hydrosystem mortality may be expressed in the estuary stage.

In NMFS analyses, the effectiveness of dam breach hinges on the degree of delayed or latent mortality that is caused by the hydrosystem and experienced by spring/ summer chinook. Yet for some spring/summer chinook populations, NMFS indicates that extinction risks may be high enough such that to “delay action and study more” carries with it a substantial risk that some populations might be lost in the interim. Therefore, for the Snake spring/summer chinook, the most risk averse action would include dam breach, a harvest moratorium, and vigorous improvements in other areas as well, habitat, hatcheries etc...(NMFS 1999).

Like PATH results, for Snake River fall chinook and steelhead, NMFS results are more clear and demonstrate that dam breaching by itself is likely to lead to recovery. However, temporary harvest reductions are also likely to reduce the short term risk of extinction for both fall chinook salmon and steelhead. NMFS notes however, that while dam breach will also increase available habitat for fall chinook (since they spawn and rear in the mainstem), harvest reductions will not affect the available habitat.

STUFA Analytical Comparison of CRI and PATH

The greatest difference between PATH and CRI results comes not from the model structure (which is very different), but from differences in the underlying assumptions about the effect of the hydrosystem on delayed mortality in the estuary and early ocean. By incorporating the delayed mortality component into the CRI matrix model, STUFA found that the relative rank of expected benefits from management actions evaluated in PATH and the modified CRI model were consistent. Dam breaching provided a much greater benefit to Snake River spring/summer chinook under a wider range of uncertainties than did the maximized transportation options.

The STUFA analysis also demonstrated that the CRI conclusion about the effectiveness of transportation-based options, that improvements short of breaching will not provide much benefit to these stocks, is consistent with PATH. In addition, the STUFA analysis also showed that both PATH and CRI are consistent in demonstrating that a reduction in harvest will not provide much benefit to spring/summer chinook, largely because harvest is already minimal.

Freshwater spawning and rearing survival.

Petrosky et al. evaluated trends in freshwater spawning and rearing survival based on the number of smolts per spawner from the 60's to the present (in press). Their analysis found no evidence for a decline in freshwater spawning and rearing survival of a magnitude that could explain the overall decline in survival of Snake River spring/ summer chinook. In contrast to the CRI results indicating a large potential for recovery based on freshwater habitat improvements, the lack of decline in freshwater spawning and rearing survival over the time period when these stocks have plummeted suggests that improvements to freshwater habitat may yield little increase in survival overall. In addition, based on these data, it appears unlikely that these improvements to freshwater habitat alone would be sufficient to halt the decline of these stocks given that a large part of the overall decline appears to have been caused by mortality associated with passage through and cumulative or delayed effects of the hydrosystem. These results are consistent with PATH results on the lack of potential for increases in survival from habitat improvements and consistent with the observation that several of the index stocks considered spawn and rear in wilderness areas with good to pristine habitat.

Delayed hydrosystem mortality

Substantial evidence supports the existence of delayed mortality for Snake River spring and summer chinook and links delayed mortality to hydrosystem experience. This evidence comes in the form of published literature, indirect evidence from life-cycle modeling, and direct evidence from fish-tagging experiments. Further, the compilation of evidence that links hydrosystem experience to delayed mortality has implications for the analyses of management options considered for the recovery of Snake River salmon and steelhead, especially the outcome of a dam breach option. In those analyses, dam breach is likely to dramatically increase the survival of Snake River chinook and lead to eventual recovery of these stocks if delayed mortality is related to hydrosystem experience. Dam breach is predicted to have less of an effect and may be insufficient for recovery if delayed mortality is unrelated to hydrosystem experience. Regardless of the action that is chosen to recover these stocks, the amount of hydrosystem mortality that needs to be compensated must include both direct and delayed components. Based on the evidence summarized in Budy et al. (*in review*), it seems implausible that little or none of the delayed mortality of Snake River fish is related to the hydrosystem.

This paper has 8 sections:

I. Introduction

II. Summary of PATH

III. Summary of NMFS CRI

IV. Discussion of the STUFA PATH CRI analytical review and comparison to PATH

V. Simple example and explanation of delayed and extra mortality, a key uncertainty.

VI. Assessing the degree of improvement in survival possible at the egg-to-smolt stage.

VII. The potential for freshwater habitat improvements; An evaluation of a set of index areas in Idaho and Oregon.

VIII. Summary of the evidence that links a fish's experience in the hydrosystem to their survival in later life stages: does delayed hydrosystem mortality exist?

IX. **Bottom Line**

I. INTRODUCTION

The depressed status of Snake River stocks of spring/summer chinook and the recent listings of many salmon stocks in the Columbia Basin have led to analytical evaluations and management advice aimed at recovery of these stocks. A considerable amount of analytical work done so far has centered around the effects of the hydrosystem and the potential for recovery and minimizing extinction if four dams on the Lower Snake River are breached, relative to a no-breach option (PATH). These analyses are focused on four endangered stocks of Snake River salmon and steelhead which must migrate past the four lower dams on the Snake River as well as the four dams on the lower Columbia River. More recently, other analytical approaches have considered improvements in survival from changes in the hydrosystem in addition to changes in the other "H"s, habitat, harvest, and hatcheries. The PATH decision analyses and these additional modeling approaches analyzed the effectiveness of management actions aimed at recovering Snake River salmon and steelhead.

Analytical evaluations of salmon and steelhead survival and the impacts from the hydrosystem date back many years in the Columbia Basin. Early models focused both on estimating the mortality of the hydrosystem based on different types of marking and transportation experiments dating back to the late 70's and on estimating the survival of fish across the life cycle. Historically, the Northwest Power Planning Council (NPPC) developed and used an early set of life cycle and passage models for evaluating hydrosystem and sub-basin planning efforts (SPM and PAM models; McConnaha and Andersen, 1992). In addition, two different groups used and presented results from their own independent models: the states, tribes and the US Fish and Wildlife Service (USFWS) developed and utilized a model now known as FLUSH, and Bonneville Power Administration (BPA) relied on and paid for the development of CRiSP, a model created by consultants associated with the University of Washington. The FLUSH model typically estimated higher mortality rates associated with dam passage and the hydrosystem, while CRiSP typically resulted in more optimistic estimates (Marmorek and Peters, 1998a).

Differing results from these models presented decision makers with conflicting management advice. The dueling models and lack of collaboration culminated in 1993 when Oregon and Idaho sued the National Marine Fishery Service (NMFS) over their ruling that the hydrosystem was causing “no jeopardy” to threatened and endangered stocks of Snake River salmon and steelhead. Judge Malcolm Marsh, in a federal district court hearing, ruled that NMFS was arbitrary and capricious in their determination of “no jeopardy” and that NMFS needed to collaborate with the states and tribes in clarifying the differences among models and hypotheses and in completing a comprehensive biological decision analysis in preparation for the next Biological Opinion for the hydrosystem due in 1999 (Marsh 1994). The collaborative group that was formed to do the biological decision analysis was called PATH - the Plan for Analyzing and Testing Hypotheses.

The PATH forum was composed of modelers, fishery biologists and statisticians from all three states (OR, WA, and ID), the federal government (Army Corps of Engineers (USACE), USFWS, NMFS, BPA, the treaty tribes of the Columbia Basin (represented by the Columbia River Inter-tribal Fish Commission -CRITFC), and the NPPC. The process was facilitated by an independent consulting firm from Canada (ESSA) and guided by three independent scientists. In addition, all PATH products and reports were reviewed by a panel of four leading independent scientists with expertise in modeling, fisheries, conservation biology, and statistics. PATH was a biological decision analysis process designed to quantitatively evaluate different actions for future operation of the hydrosystem using criteria for survival and recovery of listed Snake River salmon and steelhead. The decision analysis used a systematic approach that breaks the complex problem of determining how the operation and configuration of the hydrosystem can be improved to recover listed salmon and steelhead into its component parts. This decision analysis explicitly takes into account a wide range of uncertainties on assumptions about past stock performance, effectiveness of management actions, and future climate in the evaluation of each hydrosystem action. Decision analysis helps determine how uncertainties affect the projected likelihood of survival and recovery of listed salmon and steelhead under each hydrosystem configuration.

In PATH, the decision analysis has four components: 1) evaluation of different assumptions (hypotheses) about how environmental factors affected past survival of salmon and steelhead (retrospective analysis), 2) prospective analysis projects the number of spawning salmon and steelhead under each hydrosystem action and combinations of assumptions about past stock performance, effectiveness of management actions, and future climate, 3) estimation of the likelihood of meeting the Endangered Species Act (ESA) standards for survival and recovery of listed salmon and steelhead under each hydrosystem action based on projections of the number of spawning salmon and steelhead over a range of assumptions (risk analysis), and 4) assessment of the likelihood that certain key assumptions are true based on a comprehensive evaluation of evidence for and against each assumption (weight of evidence analysis). Given that there are limitations in data on past and present conditions, uncertainties about future conditions (like climate), and different interpretations of data, decision analysis explicitly quantifies these uncertainties and clarifies risk. Decision analysis shows which management actions are the most robust to remaining uncertainties (i.e. the least risky) and allows a decision to be made with full consideration of uncertainty and risk. PATH considers a wide range of plausible hypotheses about sources of mortality and incorporates this information into the analysis.

As the PATH forum neared completion of the biological decision analysis, NMFS initiated their own analytical effort called CRI- the Cumulative Risk Initiative. This NMFS effort did not provide for direct participation by the states, tribes, and other federal agencies. Results of the CRI analyses were periodically presented and discussed in public workshops. Involvement of other fishery agencies was based on the submission of written critiques of papers presented at these workshops. CRI is a flexible framework for providing scientific guidance regarding salmonid recovery in general, and all of the risk factors that influence prospects for recovery. CRI analyses explore the demographic effects of hypothetical reductions in mortality at different life stages based on current conditions. CRI is not a formal decision analysis model, and hence it does not deliver particular probabilities of “success” associated with particular management actions. Instead, results are first presented as the percentage increase in survival rate, or decrease in mortality rate, needed to halt the decline. For example, according to CRI, a 10% reduction in egg-to-smolt survival appears to be sufficient to halt the decline in these stocks. Then, for some management actions, the expected increase in survival from that action is then compared to what is needed. For other actions or uncertainties, changes in survival and the effect those changes might have on increasing the population growth rate are evaluated through sensitivity analyses. In these cases, results are presented as a range of possible survival values in relation to changes in population growth rate as part of "what if" scenarios or numbers experiments. For example, "what if" the delayed hydrosystem component of estuary mortality is reduced to 9, 6, 3,and 0%? Does the population growth rate increase sufficiently?". CRI does not explicitly consider past data (prior to 1980) or historical patterns (e.g. changes in environmental variation) as part of the modeling process. The feasibility of survival changes is to be considered separately in future analyses.

The CRI analysis has three components: 1) evaluation of extinction and population growth rates, 2) numbers experiments to evaluate the effect of hypothetical changes in survival at different life stages, 3) general consideration of the biological feasibility of specific actions. Based on this approach, CRI evaluates the effectiveness of general management actions that are proposed for the future and the importance of changes that have been made already. While extinction risk is a primary component of the CRI modeling framework, the risk of different management actions relative to one another and of critical uncertainties is not considered. CRI analyses are focused on increasing the population growth rate to avoid short term extinction. Apparent differences in approaches and results from PATH and CRI led to an analytical comparison performed by a sub-group of PATH called STUFA composed of scientists from OR, ID, WA, USFWS, and CRITFC (STUFA stands for states, tribes, and USFWS fishery agencies).

In general, these different analytical approaches have been used to evaluate the likely effectiveness of different management actions aimed at halting the decline of Snake River salmon and promoting their eventual recovery. The effectiveness of dam breach and other management actions aimed at recovering these fish have been essential components of several important documents aimed at facilitating decision making (e.g., USACE Lower Snake River

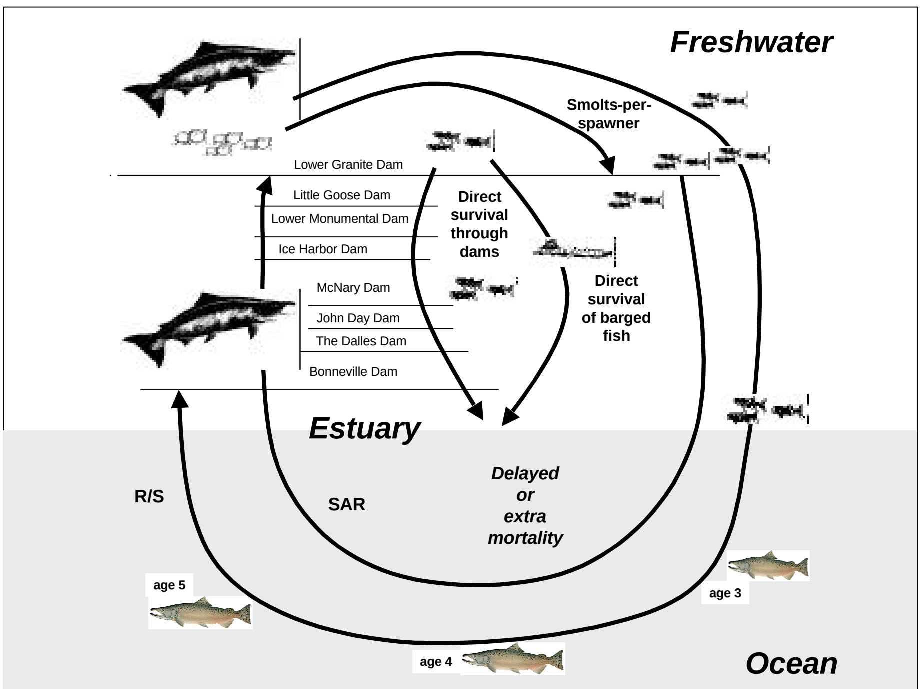
Hydro Options IS (NMFS 1999), NMFS hydrosystem operation biological opinions (NMFS 2000b), and the Federal All H paper (NMFS d)). The PATH and CRI modeling approaches and results are summarized in the following sections and described in greater detail in the A-fish Appendix A (NMFS 1999) of this EIS, in Marmorek and Peters 1998 a-b, in Peters et al. a-b, both *in press*, in NMFS 2000 a-d, and Karieva et al. 2000. The 2000 Biological Opinion on the Operation of the Federal Hydrosystem, released in December, 2000, relies primarily on NMFS CRI analyses. In addition to these analytical efforts, the Northwest Power Planning Council has sponsored a modeling framework called EDT - Ecosystem Diagnostics and Treatment. This modeling process was developed as part of their Multi- Species Framework and is used primarily as a planning tool for sub-basin planning although there is some overlap with PATH and CRI.

II. PATH

Modeling and Analytical Framework

The basic structure of PATH modeling and analysis is a life cycle model based on historical estimates of mortality at different life stages and spawner and recruit counts. The survival of salmon over their life cycle is estimated from counts of spawning salmon in freshwater streams and returning adults from these original spawners (offspring). The total survival is then compartmentalized into various life stages including 1) survival during downstream migration (smolt stage), 2) survival in the estuary and early ocean, and 3) survival during upstream migration as adults (Figure 1). For some life stages, PATH estimates mortality using either external data and/or other models that use external data to generate mortality in that life stage. For example, ocean harvest rates are based on CWT (coded wire tag) recoveries in ocean fisheries, and downstream survival is based on reach survival estimates from PIT (passive integrated transponder) tag, freeze brand, and CWT marked fish. For those life stages where there was not an external method of determining mortality, the <u>amount</u> of mortality in that life stage was first quantified by subtracting all the other known mortality in other life stages such that the remaining portion of mortality is considered ‘unexplained mortality’ (or “extra”). Then, hypotheses about that remaining mortality, and the total survival derived from the models, were compared to historical data.

Figure 1. Schematic life-cycle diagram for Snake River salmon and steelhead showing passage past the dams and the different indices of survival across different life stages. Smolts/spawner provide an estimate of freshwater and rearing spawning survival. SAR is the survival rate of smolts to returning adults (smolt-to-adult), measured at Lower Granite Dam. R/S is recruits per spawner, measured by back calculating recruits (progeny) from spawners (parents) based on redd counts, age structure, and harvest. Direct and overall survival of fish that migrated through the hydrosystem and in barges is based on PIT-tag mark and recapture experiments. Delayed or extra mortality occurs in the estuary and early ocean and may or may not be related to hydrosystem experience.



PATH modeling efforts are divided into <u>retrospective</u> modeling, to generate parameter values and basic model structure based on <u>historical</u> information, and <u>prospective</u> modeling, where stock performance is evaluated in the <u>future</u> under different management actions like dam breaching. Retrospective models used historical data on spawner and recruits, dam passage mortality, harvest, and climatic or environmental variation to account for the overall pattern of survival. Measurement, or estimation error, the error that occurs when we try and estimate model parameters from data sources, is roughly approximated and incorporated in the life cycle models. For prospective analyses, different hypotheses about the likely effect of the management actions are applied to mortality in each life stage. The <u>difference</u> between survival in the retrospective (past) period and survival in the prospective (future) period drives PATH model predictions and captures the benefit of a specific management action (i.e. tells you how much improvement you can expect from something like dam breaching vs. continued transportation).

PATH uses two different life cycle models to capture different hypotheses about mortality at different life stages.   Both models are based on Ricker spawner-recruit relationships (both assume some underlying density dependence is operating). In the first, the delta model, extra mortality that occurs in the estuary and early ocean is compared among similar stocks that must migrate past 8 dams versus those that migrate past 4 or fewer (upper river versus downriver stock comparisons). A simplified explanation of delayed and extra mortality is provided in section V. below. The second approach, the alpha model, does not rely on comparisons of upriver and downriver stock mortality but instead estimates a different extra mortality for different regions. The delta model approach relies on statistical evidence that upriver and downriver stocks share some common mortality in response to climate conditions whereas the alpha model assumes that there is no common factor that influences both stock groups.

In general, for both PATH life cycle models, estimates of total extra mortality are determined by the historical pattern of spawners and recruits and estimates of mortality rates in specific life stages (e.g. transportation or down-river migration). Both models also use the ‘D’ parameter to determine how much of extra mortality is due to the differential delayed mortality from transportation. If ‘D’ is high (there is little differential delayed mortality), then a large amount of the extra mortality is unrelated to transportation and remains to be explained. If ‘D’ is low (there is substantial differential delayed mortality), then a smaller amount of extra mortality must be accounted for.   And as noted above, in terms of the proportion of mortality that is affected by transportation, it is largely the change between historical ‘D’ values and ‘D’ values estimated for the future, and the proportion of the population that is transported, that determines the relative increase in survival under transportation and dam breach management actions.

In PATH, as part of the overall life cycle, two alternative passage models were used to predict the effect of proposed hydrosystem actions on downstream survival through the hydrosystem based on predicted flows, direct passage mortality, and information about transportation (where applicable) including ‘D’. In general, the passage models encompass a range of optimistic to more pessimistic assumptions about the way things will work in the future, under different management actions. Newer PIT tag data (which provide information about ‘D’) first discussed by NMFS in their first A-fish appendix (1999) were not available at the time PATH finalized their analyses. Debate over the value of ‘D’ based on this new PIT tag data, and implementation of ‘D’ in prospective analyses continues and is described in Bouwes et al. 1999 and summarized in general in section VI below. PATH modeled alternative hypotheses for ‘D’ and generated a range of ‘D’ values (0.12-0.86) based on output from the two passage models.

Both life cycling modeling approaches (including the 2 passage models) are used in the PATH decision analyses and reflect uncertainties (or alternative hypotheses) about the way things work and about the future. Life cycle models are run thousands of times for a given set of inputs or hypotheses in order to capture the full range of possible ways the system works and the full range of possible futures. **PATH model results show which management action has the highest probability of survival and recovery based on the range of uncertainties and different hypotheses about how management actions will affect mortality in certain life stages.**

The final stage in the PATH decision analysis is a weight of evidence (WOE) process where an independent scientific review panel completes an assessment of the likelihood that certain key hypotheses are true. Their analysis is based on a comprehensive evaluation of evidence for and against hypotheses about critical remaining uncertainties. The weight of evidence is a formal and rigorous process that does not require consensus; weights are captured using a numerical scale based on an expert elicitation process designed to minimize the influence of panel members on each other. The scientific review panel is a panel of independent experts who represent a wide range of expertise including stock assessment, general model development and analysis, population dynamics, conservation biology, hydrosystem operation and management, fish biology, general ecology, and formal decision analysis. The WOE process allows the set of assumptions or hypotheses to be narrowed down, effectively making the response under different management actions more transparent.

Historical data and base periods:

PATH separates analyses into two parts, a retrospective analysis and a prospective analysis. The retrospective analysis is designed to determine which assumptions are most consistent with historical spawner-recruit data and to create quantitative estimates of parameters needed to run models into the future. Spawner and recruit data from the early 50's to the present were used to fit a life cycle model for spring/summer chinook and from the mid 60's to the present for fall chinook. The life cycle model approach used in PATH describes the characteristics of the population that determine population growth potential (productivity) and limitations (carrying capacity) before large scale changes occurred in this system that may have altered these population characteristics (non-stationary processes). Potential benefits of future management actions are evaluated based on the full production potential and carrying capacity limitations described by the historical data.

Retrospective analyses, and comparisons of prospective models and hypotheses to historical data, allow evaluation of model fit, under different configurations and for these different hypotheses. For spring/summer chinook, models that include upstream/downstream comparisons and passage input (hydrosystem downstream survival estimates) fit the time series of Snake River spawner-recruit data much better than simple models that only consider inherent stock productivity. Comparisons of stock productivity from before the completion of the hydrosystem to after completion of the hydrosystem indicate that upriver stocks (which migrate past 8 dams) experienced a change in productivity that is coincident with the construction and completion of the hydrosystem. Based on these retrospective comparisons, PATH analyses focus on factors which are believed to have had the greatest effect on Snake River chinook since the early 1950's.

Delayed and Extra Mortality:

In general, delayed or extra mortality is the mortality of fish, from the time they leave the hydrosystem to the time they return as adults that is not accounted for by spawner and recruit relationships, direct mortality in the hydrosystem, or harvest and adult passage mortality *(see section V. below)*. This mortality includes estuary and ocean survival and can include mortality from yearly variations in the climate and environment. 'D' is a measure of the mortality of fish that were transported around the dams relative to fish that traveled in river, from the time they leave the hydrosystem to the time they return as adults. D=1 indicates there is no difference; D<1 indicates the transported fish experience a higher rate of mortality after they leave the hydrosystem. The 'D' value describes, in part, the effectiveness of the transportation program (high D = more effective, low D= less effective). Extra mortality is compartmentalized for in-river fish and fish that were transported based on 'D'. In addition, under some hypotheses, a common year effect (affects all stocks equally) due to climate and environment can be separated out from the overall mortality. When 'D' is low, much of the extra mortality is explained by an ineffective transportation system; when 'D' is high, most of the extra mortality must be explained by hydrosystem related delayed mortality of both groups or another mechanism (e.g., reduces stock viability of upriver stocks).

In PATH, extra mortality and 'D' are modeled explicitly based on retrospective and prospective comparisons. Extra mortality is modeled using the two different life cycle models described above --one which relies on the difference in extra mortality between up-river stocks and one which assumes that the extra mortality of stocks from different regions is independent. In addition, three different hypotheses are considered in prospective analyses of extra mortality; extra mortality is related to either hydrosystem experience, stock viability, or ocean regime shifts. A range of 'D' values from 0.12 to 0.86 are modeled, reflecting pessimistic to optimistic assumptions about transportation effectiveness. The 'D' value is assumed to change from year to year with different flow and management practices, both retrospectively (past) and prospectively (future).

Criteria for evaluating success:

The primary performance measures used for evaluating the management actions in the PATH decision analysis are based on the NMFS jeopardy standard for the proposed 1994-1998 operation of the hydrosystem (jeopardy to the survival and recovery of Snake River salmon and steelhead under ESA). This "jeopardy standard" has three elements. The first two elements establish benchmarks for survival. The third element establishes a benchmark for recovery. To avoid jeopardy, <u>all three elements of the jeopardy standard must be met</u>. First, PATH estimated the <u>probability</u> or likelihood that the number of spawning salmon and steelhead under a particular hydrosystem action would meet or exceed the standards for <u>survival</u> and <u>recovery</u> based on the distribution of predicted spawners in the future. PATH determined the level of <u>risk</u> associated with each hydrosystem action based on its likelihood of meeting the standard across a broad range of assumptions about future conditions.

The survival threshold is a level below which continued survival appears to be highly uncertain, due to risk factors and a lack of information regarding population responses at low spawning levels (BRWG, 1994). The benchmarks for survival were defined as an acceptable likelihood that the populations will be at or above the threshold population levels over 24 and 100 years. For the listed species, the likelihood of meeting or exceeding threshold population levels for survival must be high. In the 1995 Biological Opinion, NMFS implies that 'high' is 70% over 24 and 100 years (NMFS 1995). Depensation, deleterious population processes that occur at small population sizes (e.g. can't find a mate) were not modeled in PATH and may be responsible, in part, for optimistic projections of spawners into the future under all management actions. PATH did not explicitly model extinction risk in their analyses. Instead, they were directed to model the

NMFS survival standard in lieu of a quasi extinction level (a quasi extinction level is a level below which continued survival of the population is assumed to be precluded).

The Biological Requirements Work Group defined recovery as restoration of the “quality and quantity of the Columbia River/Snake River ecosystem … so that it can support self-sustaining and self-regulating populations of listed salmon species as persistent members of the native biotic community.” The jeopardy standard defines threshold population levels for recovery of each listed species and establishes one benchmark for recovery. Based on the approach used by the NMFS recovery team in setting the de-listing criteria for the Snake River spring/summer chinook ESU as a whole, the recovery thresholds for the individual indicator stocks was determined by taking 60% of the 1962-1967 average number of spawners for each stock. The recovery benchmark is defined as an acceptable likelihood that the populations will be at or above the threshold population levels at the end of 48 years. For the listed species, the likelihood of meeting or exceeding threshold population levels for recovery must be moderate to high. In the 1995 Biological Opinion, NMFS implies that moderate to ‘high’ is 50% after 48 years.

Implementation of Management Actions

PATH prospective analyses examined three general approaches for operating and/or configuring the hydrosystem in the future. Three actions are transportation based and use barges to move juvenile fish around dams and reservoirs. One action (A1) describes the 1995 Biological Opinion or status quo conditions. The other two actions (A2/A2’) describe improved transportation under moderate and optimistic assumptions about the extent and effectiveness of improvements to the existing hydrosystem. Three actions are natural-river (dam breach) based and remove the earthen portion of each of the four lower Snake River dams (and one Columbia River dam) and return the reservoirs upstream of the dams to natural levels under different time-lines for dam breach. Each of the three approaches is intended to reduce mortality of juvenile salmon and steelhead caused by the hydrosystem, reduce adult upstream passage mortality, and reduce the selective pressures of past management programs that reduce genetic and life history diversity.

"Transportation" and "optimum in-river migration" actions attempt to achieve these reductions through technological fixes to the existing hydrosystem. Dam breach actions attempt to achieve these results by reducing the number of dams and reservoirs salmon and steelhead must pass. In addition, for fall chinook, dam breach actions are also expected to increase spawning and improve rearing success in the mainstem and reduce adult salmon mortality caused by the hydrosystem (all species). Prospective analyses of the management actions were implemented over the full range of hypotheses concerning production potential, carrying capacity, survival through the hydrosystem and effectiveness of transportation (D), and future causes of extra mortality. In addition, the prospective analyses were conducted over a range of future climate and hydrologic conditions for the Columbia River Basin.

Sensitivity analyses to changes in habitat (improvements and further degradation), reductions or increases in harvest, and increased juvenile predation (avian predators) were used to evaluate the effect of these changes on the probability of survival and recovery. These sensitivity analyses to the other 'H's did not separate the first component of feasibility, possible biological benefit, from what was actually modeled. In other words, sensitivity analyses considered only improvements which were deemed to be physically possible on a stock by stock basis (e.g. streams which are in pristine habitat experienced no biological benefits from habitat improvement). These sensitivity analyses did not consider the second component of feasibility --the likelihood that the action will be accepted and implemented by interested parties.

Conclusions and Model Outcomes

Status of the Stocks:

Numbers of wild or natural returning spawners indicate that chinook stocks have continued to experience a severe decline in population abundance in recent years. For spring/ summer chinook, it was clear at the time of the 1993 Biological Opinion on the operation of the hydrosystem that these stocks were undergoing a continued decline and had reached dangerously low levels. Since then, numbers of returning wild spawners have continued to decline (Table 1). As demonstrated in Table 1, most of the index stocks of spring summer chinook have been below the PATH survival threshold most of the years in the recent past. Recall that the survival threshold is a low number of adult spawners that was to be avoided and below which the population is likely to demonstrate unpredictable behavior and be at a high risk of extinction.

Table 1. Depressed status of the PATH index stocks of spring/summer chinook. Stock status is shown as the number of returning wild spawners from return years 1994-99. Shaded bars show the years where the number of returning spawners for each stock was below the NMFS survival standard, a lower threshold of 150 or 300 fish, depending on stock. The survival threshold is a lower level to be avoided, below which 'bad' and 'unpredictable' things (e.g., extinction) are likely to happen. The last two columns show the average number of spawners from 1983-93 used in the 1993 Biological Opinion (when dam breach was first discussed because of depressed stock status) compared to the even lower average spawners for recent years (94-99).

STOCK	SURVIVAL THRESHOLD	1994	1995	1996	1997	1998	1999	83-93 BiOp AVG.	94-99 AVG.
Bear Valley	300	33	16	56	225	372	72	384	129
Marsh Creek	150	9	0	18	110	164	0	182	50
Sulphur Creek	150	0	4	23	43	140	0	162	35
Johnson Creek	150	50	23	58	236	121	49	NA	90
Poverty Flat	300	209	81	135	363	396	153	494	223
Imnaha River	300	52	54	143	153	90	56	529	91
Minam River	150	16	28	223	141	121	91	243	103

Retrospective (Past):

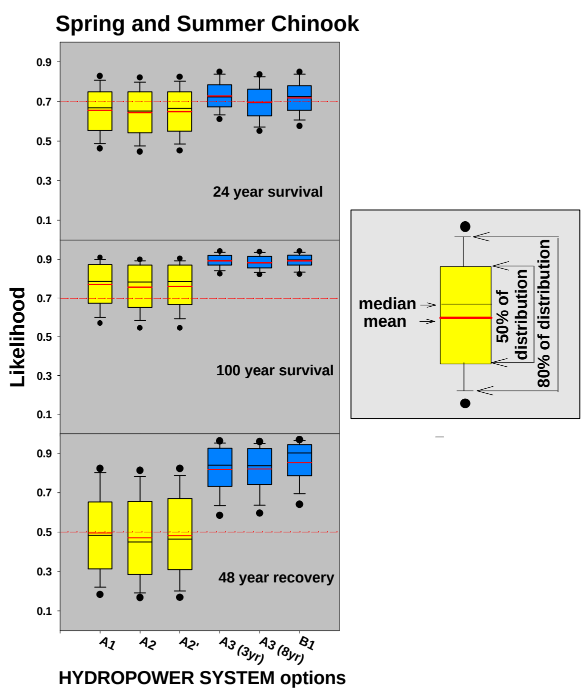
PATH retrospective analyses led to several important conclusions which directed prospective modeling. Declines in numbers of spawning salmon in the Snake River Basin and upper Columbia River Basin populations were greater than for lower Columbia River Basin populations. The declines in Snake River Basin populations after 1974 were greater than declines before 1974, and greater than declines in lower Columbia River Basin populations after

1974. 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 hydrosystem. The significant decline in Snake River Basin populations after 1974 does not coincide with habitat degradation. Most habitat degradation occurred prior to 1974, and spawning and rearing habitat for a number of the Snake River Basin populations has remained in good or pristine condition. The degree to which artificial propagation contributed significantly to declines in survival of listed Snake River Basin populations is uncertain. Declines in survival of Snake River Basin populations occurred at the same time as substantial declines in harvest rates. 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. All models suggest that the mortality below 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 (i.e. D <1) for Snake River spring and summer chinook.

Prospective (Future) Results:

Prospective PATH analyses for spring and summer and fall chinook salmon indicated that the dam breach actions had higher probabilities of achieving the survival and recovery standards than the "transportation" actions (Figure 2). Dam breach actions met the standards over a wide range of assumptions (i.e. these actions are robust to remaining uncertainties). In fact, the dam breach actions met the 100-year survival and the 48-year recovery standards for spring/summer chinook even under the most pessimistic set of assumptions. The dam breach actions were also less risky than the "transportation" actions (i.e. model projections had relatively little variability over the full range of assumptions). For steelhead, a hydrosystem action was judged likely to meet survival and recovery standards if it meets the standards for Snake River spring and summer chinook salmon. Based on this comparison, the dam breach actions appear to have the highest probability of recovering steelhead. PATH analyses did not address whether the survival and recovery standards for steelhead would be met if a hydrosystem action fails to result in an acceptable likelihood of survival and recovery for spring and summer chinook salmon. However, recent estimates for Snake River steelhead D values were between 0.56-0.58 (NMFS, 2000b), indicating a high rate of transportation based delayed mortality.

Figure 2. PATH results for spring/summer chinook shown as a distribution with box whiskers denoting the range and percentiles of the results for each management action. A longer box and whiskers indicates less certainty (i.e. a wider range of results). Results are shown for the three NMFS jeopardy standards (see text for explanation) and the six management actions (*see text*; Marmorek and Peters 1998). Lighter boxes on the left (A1-A2') denote status quo and transportation based actions while darker bars denote dam breach options (A3-B1). To avoid jeopardy, all three standards must be met. For the average or median result, only dam breach meets all three jeopardy standards. In addition, the dam breach results are the least risky, as demonstrated by the tighter box and whiskers.



Results for fall chinook under dam breach are more certain than for spring and summer chinook because this stock benefits not only from increased juvenile survival during downstream migration, but also from increased spawning habitat (carrying capacity) in the mainstem. Spring and summer chinook spawn primarily in smaller tributaries and streams while fall chinook historically spawned in the mainstem Snake River areas, which are currently inundated. The increase in carrying capacity for fall chinook under dam breach is independent of any transportation assumptions and could <u>not</u> be achieved with any other management action.

Habitat:

For PATH prospective model runs of spring and summer chinook, the effects of habitat improvements were addressed through a sensitivity analysis where improvements to habitat were expressed as an increase in stock productivity and were assumed to represent all possible improvements to the freshwater habitat. This approach was used to look at how various assumptions about future changes in spawning and rearing habitat conditions would affect the results of the various hydrosystem management actions. The habitat sensitivity analysis confirmed, however, that while habitat improvements may affect the ability of a hydro management action to meet the jeopardy standards for some stocks, the relative ranking of management actions remains the same with dam breach demonstrating the highest probability of recovery and survival overall. Thus while it is likely that habitat improvements for some spring and summer chinook and steelhead stocks will increase the productive potential in the Snake River Basin, it <u>does not</u> appear that <u>habitat improvement alone</u> would be enough to recover these stocks.

Avian Predation:

PATH also performed a detailed sensitivity analysis to a range of additional predation mortality rates (5-40%) meant to reflect avian predation in the estuary. Additional predation did <u>not</u> change the rank of management actions as all actions are affected essentially equally –the dam breach actions still show the greatest probability of survival and recovery. Further, the BKD or ‘here to stay’ hypothesis for extra mortality can also be used as a surrogate for bird (or other) predation and/or negative hatchery effects. PATH model results demonstrate that even if a large component of the extra mortality is ‘here to stay’ (as modeled under the BKD hypothesis), dam breach still has the highest probability of recovering Snake River spring and summer chinook and fall chinook and is the most robust to uncertainties.

Harvest:

The existing harvest schedules for spring and summer chinook were used in PATH prospective model runs, and like habitat improvements, the sensitivity of model predictions to future changes in harvest was also evaluated through a sensitivity analysis. Harvest sensitivity analyses included two more conservative in-river harvest rate scenarios than the one based on current management --harvest rates were reduced by 1/3 current value and also set to 0. The ranking of management actions was <u>unaffected</u> by these more <u>conservative harvest schedules</u>. Given the existing low levels of harvest on spring/summer chinook and the insensitivity of PATH model projections to harvest reductions, it is unlikely that additional future reductions in harvest alone will significantly increase the survival of spring/summer chinook such that jeopardy standards will be met.

In contrast to spring and summer chinook, Snake River fall chinook are harvested at a much higher rate in both the ocean (by both the U.S. and Canada) and in-river. Both adult in-river harvest and cumulative ocean harvest rates have fluctuated around 30% from 1965 to the present. In-river harvest of Snake River fall chinook is currently determined by river-mouth returns of both Snake River and Hanford Reach stocks (up-river bright chinook), while ocean harvest is determined largely by overall abundance and management agreements between Canada and the US originally set by the Pacific Salmon Treaty in 1985. PATH included several different harvest reduction scenarios in model predictions: 15% reduction in ocean harvest combined with the existing in-river harvest schedule, and a 50 and 75% reduction in ocean harvest combined with an extremely low conservation level harvest schedule for in-river fisheries. These dramatic harvest scenarios do not change the rank of management actions –the natural river dam breach actions result in probabilities of recovery near ~100% while the transportation based management actions fail to meet the 50% recovery standard under any combination of in-river and ocean harvest reductions except the most extreme. If a 75% reduction in ocean harvest is combined with a 50% reduction in in-river harvest (that also requires recovery to be met before any non-subsistence level harvest can occur), then the transportation management action barely meets the 50% recovery standard. However, these extremely dramatic changes would have to be in place forever, and as soon as harvest restrictions were relaxed, the fall population would likely drop below recovery levels again. Again, this is in contrast to dam breach, which has a 100% probability of meeting the recovery standard for fall chinook under any of the future harvest assumptions.

III CRI:

Modeling and Analytical Framework

CRI is composed of two primary types of models, simple Dennis-type models used to assess extinction risk, and Leslie matrix models used to explore life stage effects of hypothetical reductions in mortality based on current conditions (NMFS, 1999; NMFS, 2000 a-c). The Dennis type model was based on a simple linear model of growth rate for census data, for spawner and recruit counts, for 1980 to the present (Dennis et al., 1991). This type of model requires little detailed data and is broadly applicable across species and ESU's. Early renditions did not incorporate age data (NMFS, 1999), but later versions incorporated a running sum of spawners to dampen the effects of sampling error and age-structure cycles (NMFS, 2000 c). Based on this approach, NMFS was able to estimate the average population growth rate and the probability of extinction. The risk of extinction was first estimated based on a quasi-extinction threshold of 1 fish per any year and later estimated based on absolute extinction of one fish returning for an entire generation (NMFS, 1999; NMFS, 2000 c). NMFS notes that use of this type of approach requires that the data meet three important assumptions: 1) year to year variation in spawner counts is due to environmental variability and not sample error, 2) the rate of population change (decrease) can not be changing, and 3) population changes are density independent (the number of fish alive has neither positive nor negative effects on the survival of the brood). Salmon data obviously violate all three assumptions to some degree, but NMFS suggests the violations for this time period are minor enough to have little effect.

NMFS used a Leslie matrix model to evaluate management options based on data from brood years 1980 to the present, and 1990-94, depending on the question, for estimating population parameters for age-structured matrices (NMFS, 1999; NMFS, 2000 a-c; Kareiva et al., 2000). Analyses were restricted to recent years because these types of matrices are limited to data input over stable time periods, and these stocks have experienced a progressive decline in productivity (a non-stationary change) over time (e.g. the rate of decline is increasing). The matrices isolate survival for life stages representing: 1) egg-to-smolt, 2) downstream smolt, 3)estuary and ocean, and 4) upstream adult (Figure 1). Like PATH, mortality at some life stages (e.g. downstream survival) was estimated based on external data or sub-models (e.g. SYMPASS passage model that relies on PIT tag marked fish and roughly averages PATH FLUSH and CRiSP passage models). Egg-to-smolt survival was estimated after all other sources of mortality had been estimated, as a residual of remaining mortality. These matrix models were density independent, as NMFS found little evidence supporting density dependence for the recent time period of data used. Extinction models of some form can be broadly applied to salmonids (if a constant average age structure is used), but life-table matrices are possible for only some ESUs (evolutionary significant unit) where data are plentiful (since many populations lack age-structured data).

For evaluating potential management options, numerical sensitivity experiments are conducted to see what hypothetical changes in survival estimates elicit the greatest improvement in annual population growth, to the extent that data allow it. This mathematical exercises represents “what if experiments”. For example, “what if we could increase downstream survival of salmon? How much would that increase annual rate of population growth?” The point of these experiments is to direct attention towards life stages that provide the greatest opportunities for reversing population declines, based on NMFS estimations of how mortality is distributed across the life cycle. They identify the stages at which research should be directed, as well as estimating the magnitude of change necessary at various life stages in order to halt declining trends. Uncertainty is expressed in a crude verbal scale of “highly likely”, “likely”, and “possible”. Historical data may be consulted for particular questions, but historical data are not the focus of CRI analyses.

As the final step and using the guidance from the “what if” experiments, CRI pursues feasibility studies. These typically proceed from general explorations to specific management actions. For example, to consider the feasibility of habitat improvement as a means of fostering salmonid recovery, CRI begins by asking broadly whether one can find statistical evidence linking salmon productivity to habitat features. Lastly, CRI plans to investigate the feasibility of actually implementing selected habitat improvements, for example (could we really get those reduced sediment loads?) and to imagine coordinated sets of actions and their effects on all twelve threatened or endangered ESUs in the Columbia Basin. These final steps have not been fully completed.

Historical data and base periods:

As mentioned above, CRI does not perform retrospective analyses, but instead starts by defining current conditions under the status quo. For the Snake River ESUs “current conditions” has been defined to be 1980 onward, or 1990 onward depending on the analysis on hand. The 1980 data was selected because it marks a period by which many of the major improvements in the hydropower system had been completed. The 1990-94 data were chosen to avoid steady declines in productivity. CRI views “current conditions” as the starting point, and the assignment of “blame” for past declines in numbers to be of secondary importance. For example over harvest may have contributed substantially to the decline of Snake River spring chinook salmon; however, the trends in declines of Snake stocks continued despite severe harvest restrictions. Therefore under current conditions further restrictions of harvest offer little possibility for recovering this ESU. In analyses to date, CRI has not incorporated positive or negative effects of ocean cycles, and what are called “Pacific decadal oscillations”, but this is a high priority for CRI research. In later analyses, however, CRI does use the matrices to test the effectiveness of past management actions: 1) reductions of harvest rates, 2) engineering improvements that increased juvenile downstream survival, 3) the transportation program (Kareiva et al., 2000).

Delayed and Extra Mortality:

In working with PATH models NMFS has in the past used delayed and extra mortality measures, and there historically were substantive differences among groups regarding the best estimates of what D-values (transport extra mortality) are under current transportation systems. However, in the new and more general CRI analytical framework “differential delayed mortality” and “extra mortality” are not part of the analytical framework. The reasons NMFS provides for reformulating the question in a way that avoids the D and extra-mortality jargon include: 1) D, or differential delayed transportation mortality, and extra mortality are most important for Snake River salmonids, not other ESU’s, 2) the detailed data required to estimate these parameters may not be available for stocks other than than the Snake, and 3) CRI feels that running the scenario of “what if dams are breached” can be done in an alternative that does not incorporate delayed mortality.

Specifically, to conduct a “*what if dams are breached*” scenario analysis, CRI increases upstream survival, downstream survival, and survival in the estuary and ocean by varying amounts, and determines how much the annual rate of population growth is increased as a consequence.   It then aims to ask, as a separate exercise, what data indicate such increases might be realized from dam breaching. CRI notes that none of these feasibility analyses are clear cut, and because the “perfect experiment” has not been designed (with and without dams, all else identical) all of these statistical analyses are tenuous.   In the future CRI intends to explicitly examine the biological mechanisms underpinning delayed mortality by looking at fitness changes of individual fish as a function of their history.

Criteria for Evaluating Success:

CRI favors three measures of success, only two of which have been applied to date, all of which are applied to "populations" or "stocks".   As recovery goals are developed for each ESU (the job of Technical Recovery Teams), there will need to be a translation of these population-level measures to ESU or "province level" goals. Whatever translation is made, the populations will still be a focus of monitoring and evaluation, and the three "population-level measures" listed below will play a central role. CRI's has chosen to emphasize measures of success that are conducive to short-term monitoring. In theory, as data from each year's counts of spawners are added, an updated estimate of annual population growth and extinction risk can be calculated, and in this way the success of management can be continually re-evaluated.

The first measure is the average <u>annual rate of change in population size (lambda)</u> when populations are at low densities and are unimpeded by competition for limiting resources. This measure is descriptive, requires no assumptions about future population responses under different management actions, and was used because it is easy to calculate from sparse data, is robustly associated with "population resilience" (or resistance to extinction), and is a metric that could be extracted from almost any population model. However, the estimate of lambda can be highly sensitive to the methodology used.

The second measure is the <u>probability of extinction</u>. Regardless of the model used to calculate this risk, probability of extinction declines with increasing population size, decreasing year-to-year variation, and increasing trends in abundance. CRI seeks extinction models of varying levels of complexity, so that decision-makers and scientists can readily adjust time horizons or risk thresholds to meet their own needs.   In practice it is "quasi-extinction" that is modeled, where quasi-extinction refers to some critical low threshold (such as 1 spawner, or 20 spawners). One problem with extinction modeling is that it typically produces extremely large confidence intervals (or levels of uncertainty); for this reason confidence intervals must be reported for any extinction risk evaluation. Confidence intervals over shorter horizons (such as 10 years) are typically much smaller than confidence intervals over 100 year time scales. As data accumulate, and improved methods for predicting population growth rates are developed, confidence intervals will narrow.

The last measure of success is long-term equilibrium population size in the absence of harvest (or carrying capacity). CRI has not used this measure yet because it requires methods for estimating density-dependence and hence carrying capacity in a variable environment -- these methods are still under development. Moreover, the contrasts in population densities available for the period that begins with 1980 (which is what CRI refers to as the beginning of "current conditions") are so limited that they make it very difficult to estimate density-dependence and hence carrying capacity. The CRI is investigating new statistical methods, as well as exploring the possibility of using habitat attributes to assess carrying capacities.

Implementation of Management Actions

CRI entails a "broad brush" and then more specific simulations of management actions. The broad-brush approach simulates the effects of potential management actions simply by changing stage-specific survival rates in the baseline matrices that represent current conditions. Since each matrix can be compactly summarized as yielding an "annual average rate of population growth", the effects of management can be concisely portrayed as having particular effects on rates of population growth. For example, one might conclude that a reduction in harvest increases population growth by 40%. In many cases, data are adequate for only a broad-brush approach, and NMFS believes that often all that is needed is a "broad brush" approach. For example, by applying it to spring/summer chinook salmon, one sees that total cessation of harvest on that ESU would only increase population growth by 1%, an increase too negligible to help much with respect to mitigating extinction risks (unless there are dozens of other simultaneous actions, all giving 1% incremental gains).

Because it is not a formal decision theoretical analysis, CRI does not examine management options in terms of which action most increases the relative probability of particular "success metrics". Instead the CRI asks "cruder" questions such as: 1) how likely is it that dam breaching by itself will rescue salmonids from the threat of extinction?, 2) are there collections of actions not involving dam breaching that have chances of success, 3) what is the evidence that every possible management action in favor of salmon should be pursued if risks of extinction are to be mitigated? In addition, NMFS later pursued additional scenarios including: 1) the effect of eliminating all direct migration mortality except for a small tribal harvest, 2) the effectiveness of past reductions in harvest, improvements to the hydrosystem, and the transportation program, and 3) the effect of improved survival in the egg-smolt and estuary/ early ocean stage in reversing the decline (Kareiva et al., 2000).

A more detailed simulation of management actions can also implemented by altering survival rates in baseline matrices and making some assumptions about environmental variability in the future and resource limits to population growth. Using these assumptions CRI generates a full spectrum of population trajectories into the future, and can then describe those results as a probability distribution (mean and variance) of population sizes at any time. Thus far such detailed simulations have only been conducted for reductions in harvest on fall chinook salmon.

CRI - main conclusions:

Status of the stocks:

NMFS has been continuously updating their extinction risk analyses since the release of the A-Fish Appendix in 1999 (Tables 2-3). In recent analyses reported in the 2000 Biological Opinion, NMFS estimated that population growth rates (lambda) were below one for almost all of the Snake River spring/ summer chinook index stocks with lambdas ranging from 0.79-1.00 (NMFS, 2000a). A lambda less than one indicates the population is declining; a lambda = 0.9 indicates that ½ of the population will be gone in 7 years. The risk of extinction for these stocks was

Table 2. Population growth rates (e.g., lambda's) reported by NMFS through time.

Stock group	**Afish Dec, 1999**	**12 ESUs 1st draft, July, 2000**	**Biological Opinion Aug, 2000**	**12 ESUs 2nd draft, Nov, 2000**
Snake River spring and summer chinook				
ESU or aggregate	NA	~0.97	NA	0.96
Bear Ck	1.116	~0.98	NA	1.02
Imnaha R	1.111	~0.95	0.79	0.93
Johnson Ck	1.17	~0.99	1	1.01
Marsh Ck	1.092	~0.95	0.94	0.99
Minam Ck	1.073	~0.94	0.80	0.99
Poverty Ck	1.184	~0.99	0.98	1.01
Sulphur Ck	1.144	~0.98	0.97	1.04
Snake River Fall chinook, ESU	NA	~0.93	0.84	0.94
Snake River Steelhead, ESU	NA	~0.96	~.50	0.91

moderate to high for about half of the stocks in the short term (24 years) and extremely high in 100 years (as high as 100% chance). For Snake River fall chinook, lambda was 0.84, and the risk of extinction was 0 in the short term but 100% in the long term. For Snake River steelhead, lambda was ~0.50, and the risk of extinction was 100% in both the short and long term (NMFS, 2000a). These estimates of the risk of extinction, however, had large confidence intervals (or little certainty) over the short term. Regardless, these analyses demonstrated that a substantial increase in the population growth rate (lambda) is required to reduce the risk of extinction to an acceptable level.

Table 3. Short term extinction risk (over 10 or 24 years) reported by NMFS through time. A value of 1 means there is a 100% chance of extinction over that time period; a 0 indicates there is no chance of extinction.

Stock group	**Afish Dec, 1999 10 yrs**	**12 ESUs 1st draft, July, 2000**	**Biological Opinion Aug, 2000**	**12 ESUs 2nd draft, Nov, 2000**
Snake River spring and summer chinook				
ESU or aggregate	NA	NA		0
Bear Ck	0.01	mean 0.16	NA	0
Imnaha R	<0.0001	mean 0.16	0.23	0
Johnson Ck	0.0009	mean 0.16	1.00	0
Marsh Ck	0.15	mean 0.16	0.01	0
Minam Ck	0.04	mean 0.16	0.48	0
Poverty Ck	0.0002	mean 0.16	0.00	0
Sulphur Ck	0.10	mean 0.16	0.13	0.05
Snake River Fall chinook, ESU	<0.0001	NA	0.00	0

In addition to their original analyses, NMFS later completed their large-scale multi-species risk assessment (NMFS, 2000c), which also estimated lambdas and extinction risk but also included another index of population status –the percent or chance of a 90% decline over 24 and 100 years. The results changed substantially relative to the results reported in the Biological Opinion with population growth rates (lambda) estimated to be near or greater than 1 for six of the seven Snake River spring/ summer chinook index stocks and lambda = 0.96 for the Snake River spring/summer chinook ESU aggregate. When lambda is greater than one, the population growth rate is supposedly increasing. Likewise, the risk of extinction was estimated to be substantially less in NMFS later analyses with a 0-5% of extinction for all of the index stocks in the short term (Table 3; 24 years) and a 0-78% risk of extinction in 100 years.

For the Snake River spring/summer chinook ESU aggregate, the risk of extinction was 0% in the short and long term (Table 3). Similarly, for the new index, the percent chance of a 90% decline over 24 and 100 years, NMFS estimated a 0 and 91% chance for the Snake River spring/summer chinook ESU aggregate and a range of 1-33% and 7-100% chance of a 90% decline for the 7 index stocks over 24 and 100 years, respectively. For Snake River fall chinook, lambda was 0.94, and the risk of extinction was 0 and 40% in the short term and long term, respectively. For Snake River steelhead, lambda was 0.91 (previously 0.50), and the risk of extinction was 48 and 100% for the short and long term, respectively (NMFS, 2000a). In contrast to earlier analyses, these later analyses suggest that <u>much less may need to be done to recover these stocks</u>.

Relative to earlier analyses, the population growth rates now appear to be mostly above one indicating the populations are now deemed to be increasing and the risk of extinction has changed from low to no chance of extinction. Some of these differences between current and past analyses are because of changing methodolgy (e.g. incorporating the effects of hatchery fish reproductive success). Additional documentation describing NMFS various extinction analyses and changes to these analyses can be found in NMFS 1999, NMFS 2000 a, b, and c. Because the NMFS estimates of extinction are continuously changing, and the pattern among the stocks has changed substantially, it is difficult to summarize NMFS assessment of current stock status. For example, the Johnson Creek spring and summer chinook index stock changed from a 100% probability of extinction to <u>no chance</u> of extinction (24 years), from analyses in the 2000 Biological Opinion to more recent analyses (Table 3; NMFS, 2000 b and c). In addition, the NMFS most recent estimates, which suggest that the stocks are mostly experiencing population growth rate increases and have little chance of extinction, does not correspond to the apparent status of the stocks based on run size (Table 1). These data, in contrast to the NMFS estimates, demonstrate that the seven index stocks have been below the survival threshold for most of the past 6 years.

Past management:

Although CRI is not based on historical data or comparisons, they did investigate the effectiveness of past management actions for Snake River spring/summer chinook including: 1) reductions of harvest rates from about 50% in the 50-60's to less than 10% in the 90's, 2) engineering improvements that increased juvenile downstream survival rates from 10% after the dams were completed to 40-60% in most recent years, and 3) the transportation of 70% of the fish from the uppermost dams to below Bonneville dam. From these analyses, NMFS determined that if these improvements had not been made, the rates of decline may have been as high as 60% and Snake River spring/ summer chinook could have already gone extinct. They further concluded that while past management attempts have reduced in-river migration mortality, they have not halted the population decline (Kareiva et al., 2000).

Future Management:

For prospective analyses, or consideration of possible future management actions, NMFS evaluated the effect of eliminating all direct migration mortality except a small tribal subsistence harvest. This direct mortality includes only mortality that occurs immediately in the hydrosystem from turbines, bypass, and ladders etc... but does not include delayed or latent mortality that may be caused by the stress of hydrosystem passage or transportation but occurs later, in the estuary and ocean. NMFS points out that while perfect (0% mortality) in-river survival is not possible, this scenario was meant to represent the intent of improving in-river survival through dam breach and/or modifications to existing structures. This analysis showed that even with 0% mortality in river for juveniles and adults, stocks would continue to decline (Kareiva et al., 2000). From these types of scenarios NMFS concluded that by itself, dam breaching is unlikely to recover spring/summer chinook salmon.

All life stages:

In addition, NMFS evaluated the effects of improvements in survival at other life stages (other than in-river migration) and whether these improvements would be enough to halt the decline in the absence of dam breach. They first concluded in the draft A-Fish Appendix (NMFS, 1999) that recovery of Snake River spring/summer chinook is likely to require dam breaching <u>and</u> many other aggressive management actions. This conclusion was based on a conservative estimate of the benefits of dam breaching, the expected benefits of habitat improvements and hatchery modifications, and does not include hypotheses about ocean conditions. They also concluded that no <u>single</u> or <u>simple</u> actions have any substantial likelihood of recovering spring/summer chinook salmon. Later analyses considered combinations of egg-to-smolt survival and early estuary and ocean survival that would halt the decline in the population (KAREIVA et al., 2000). These analyses showed that egg-to-smolt mortality must be reduced by 11% or early ocean /estuarine mortality by 9%. Further, when improvements are made that increase survival in both life stages, the <u>total</u> improvement in survival needed from <u>both</u> is less.

The primary management actions identified as having potential to increase egg-smolt survival and early ocean/ estuary survival include habitat restoration and dam breach respectively. Conclusions about the effectiveness of dam breaching, however, are limited by critical uncertainties regarding latent impacts of dams on fish fitness and survival outside the hydropower system.   Conclusions about the benefits of management actions aimed at hatcheries and habitat are hampered by an absence of the appropriate feasibility studies. NMFS indicates from their analyses that if that indirect or delayed mortality (latent impacts) is 9% or higher, dam breach would reverse the population decline of Snake River spring/ summer chinook (KAREIVA et al., 2000). Thus the effectiveness of dam breach hinges on the degree of delayed or latent mortality that is caused by the hydrosystem and experienced by spring/ summer chinook. Yet for some spring/summer chinook populations, NMFS indicates that extinction risks may be high enough such that to “delay action and study more” carries with it a substantial risk that some populations might be lost in the interim. Therefore, for the Snake spring/summer chinook, the most risk averse action would include dam breach, a harvest moratorium, and vigorous improvements in other areas as well, habitat, hatcheries etc...(NMFS, 1999).

For Snake River fall chinook and steelhead, NMFS results are more clear and demonstrate that dam breaching by itself is likely to lead to recovery. However, temporary harvest reductions are also likely to reduce the short term risk of extinction for both fall chinook salmon and steelhead. NMFS notes however, that while dam breach will also increase available habitat for fall chinook (since they spawn and rear in the mainstem), harvest reductions will not affect the available habitat.

IV. SUMMARY OF THE STUFA PATH AND CRI ANALYTICAL REVIEW.

Introduction

In response to the release of NMFS new CRI results, which in some cases appeared to conflict directly with PATH results, a sub-group of PATH called STUFA composed of scientists from OR, ID, WA, the USFWS, and CRITFC (STUFA stands for states, tribes, and USFWS fishery agencies) completed a detailed analytical comparison of these two different approaches to evaluating management options for the Snake River stocks (STUFA 2000). That analysis and accompanying report is attached as Appendix A and summarized in general below. The key results that conflicted generally revolved around the effectiveness of dam breach and habitat improvements for spring and summer chinook. In general, CRI concluded that dam breach is both unnecessary and unlikely to create a large enough improvement in survival to recover Snake River stocks. In contrast, PATH concluded that dam breach was the only management action that met all the survival and recovery thresholds and was also the least risky when the various sources of uncertainty were considered.

In addition, CRI results indicated that potential survival increases from improvements to habitat in the egg-smolt stage would be sufficient to reduce the threat of extinction to acceptable levels.

PATH results, however, demonstrated that there were few feasible combinations of habitat improvements that could increase survival sufficiently unless the dams were breached, and habitat improvements did not alter the rank of management actions (e.g. dam breach still had the highest probability of survival and recovery and the least risk). Therefore, in order to clarify differences between CRI and PATH approaches and results, the STUFA group completed a detailed analytical review of CRI and comparison to PATH. Their analysis was centered around the matrix model NMFS used to evaluate management options and the data and results for spring/summer chinook, where the greatest differences in results lay.

Key differences between PATH and the CRI matrix model

Detailed descriptions of the two different approaches are provided above and can be found in NMFS 1999. PATH modeling was performed in an open, inclusive, and independently reviewed and facilitated forum composed of scientists from the primary fishery management agencies and stakeholders in the Columbia Basin. CRI modeling is done by NMFS scientists and approaches and ideas are periodically shared with the public and other fishery agencies at workshops. Structurally, the models used by the two different approaches are very different. The PATH model is structured as a stochastic (includes variability and uncertainty) life cycle model that is run many thousands of times with different combinations of passage survival, alternative hypotheses about delayed mortality, hypotheses about the effectiveness of management actions, and variability in input parameters. The PATH modeling is part of a formal and rigorous decision analysis that provides the relative success or probability of different management actions as well as the risk associated with each. In contrast, the CRI model is a much simpler model based on life table data and a matrix approach that is used for asking general questions about how reductions in mortality at different life stages affect the population growth rate.

In terms of model results, the key differences between the two modeling approaches relate to the treatment of multiple hypotheses and uncertainty, how management actions are evaluated, and whether or not and to what degree experience through the hydrosystem is related or <u>linked</u> to survival in the estuary and ocean (delayed mortality). The PATH models explicitly incorporate uncertainty and use multiple hypotheses when data are lacking. Thus the PATH results are part of a risk or decision analysis such that results reflect a distribution of possible outcomes and the risk (or chance of being wrong) of each action. For example, model output includes the hypotheses that delayed mortality is and is not related to hydrosystem experience and shows which option is best in the face of this critical uncertainty. CRI does not incorporate uncertainty or multiple hypotheses explicitly in the model but does sensitivity analyses. These sensitivity analyses address questions about uncertainties such as how the effectiveness of dam breach changes as the amount of delayed mortality from the hydrosystem increases, for example.

In PATH, management actions and uncertainty surrounding the effectiveness of management actions are explicitly modeled with feasibility included in the expected benefit of the action. For example, in PATH, only feasible changes in habitat are included and stocks which spawn and rear in wilderness areas do not experience any benefits in egg-to-smolt survival from habitat restoration. In CRI, management actions are roughly evaluated using "numbers experiments" where mortality is reduced in various life stages and related to a hypothetical management action. For example, CRI determines which life stage gives the biggest bang for the buck in terms of overall survival increases (e.g., egg-to-smolt survival), and then relates that life stage to a hypothetical management action that occurs in that life stage (e.g., habitat improvements). Feasibility analyses have generally not been included in CRI matrix analyses but are a NMFS goal for the future. Finally, in PATH, a fish's experience through the hydrosystem is related to their survival later in the estuary and ocean through three hypotheses: 1) all or most of the extra mortality that Snake River fish experience relative to lower river stocks would be eliminated if the dams were breached, 2) none of the of the extra mortality that Snake River fish experience relative to lower river stocks would be eliminated if the dams were breached and 3) changes in the extra mortality is mostly related to ocean cycles. In CRI primary model runs, a fish's experience through the hydrosystem is <u>not</u> related to their survival later in the estuary and ocean. However, NMFS does some sensitivity analyses where they evaluate the effects of reducing estuary and ocean mortality on lambdas, or population growth rates. This sensitivity analysis is

NMFS representation of relating hydrosystem experience to delayed mortality beyond the hydrosystem.

Although they differ in technical detail, all PATH and CRI criteria for evaluating success can be conceptually related. In some cases, they can actually be converted from one into the other by direct calculation (both can produce a population growth rate or lambda), and in other cases, the two analytical approaches are sufficiently different that translation is not straightforward. Whereas CRI focuses on extinction risks, PATH focuses on survival criteria. The two are not mathematically interchangeable, but conceptually they both represent some lower threshold to population size that is to be avoided. PATH focuses more on recovery criteria than do current renditions of CRI. In addition, the management action or combination of actions that is predicted to have the greatest effect, and/or the rank of management actions can be compared regardless of how success was measured.

Objectives of the STUFA evaluation of CRI

The overall goals of the STUFA analytical evaluation were to: (1) investigate and evaluate the structure of the CRI matrix model, the assumptions used in it, and the effects of the model structure and assumptions on the distribution of mortality; (2) compare analytical methods and results of the CRI matrix model to PATH based on empirical information utilized by PATH and variations on the CRI matrix model and (3) evaluate conclusions indicated by these modifications. The STUFA group used a stepwise approach to make modifications to the CRI matrix that allowed comparison to PATH, changing as little as possible in the basic matrix, in order to make clear how each modification impacts the CRI analyses and conclusions. In addition, they created a version of the CRI matrix model that explicitly incorporated delayed hydrosystem mortality assumptions, a critical uncertainty that determines, in part, the effectiveness of dam breach used for modeling other critical uncertainties (e.g., estuary and ocean survival). Finally, they created a matrix model that could be used both to model the alternative management actions modeled by PATH (and discussed in NMFS 1999 and 2000 a, b, and c) and to evaluate the relative impacts of natural variation and uncertainty in the model input parameters.

Dennis- Holmes extinction model

NMFS's estimates of extinction probability and survival improvement needed are driven by the estimate of lambda 'ë', which is the 'finite' long term annual growth rate of the population. If ë is > 1.0, population is expected to grow; if ë is < 1.0, population is expected to decline; if ë is= 1.0, population is expected to be stable at the present spawning level. Since ë is a prediction of the rate of change of population size each year, the expected size of the population after a given number of years can be predicted, given an estimate of starting population size. For example, if ë is estimated to be 0.95, the decline in mean population size after 15 years would be about 0.5. This means in 15 years the population would be about half the size of where it started. In contrast, if ë is estimated to be 1.05 the population would roughly double its size in 15 years.

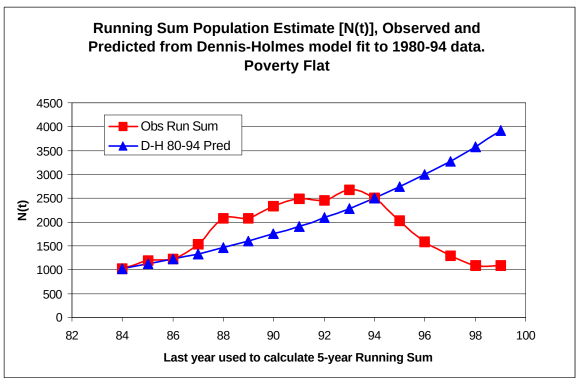
USFWS has documented a number of concerns with NMFS's current method for estimating lambda, which provides an optimistic view of stock status (USFWS, 2000). They were also concerned that NMFS is using an extinction threshold of 1 fish over five years, which yields the lowest extinction probabilities. Thus, in addition to the STUFA evaluations of the NMFS matrix model, USFWS has also done some analysis of the Dennis-Holmes model used to estimate the probability of extinction in the latest draft of the Biological Opinion. In general, the approach they used tested the predictive ability of the model by pretending as if some of the spawner data which we have observed (for example, from the last 5 years) hasn't been collected yet. USFWS did this by using the CRI method to fit the model to data from a shorter time period (estimates of spawner numbers from 1980-94, instead of 1980-99). They then compared the predicted trajectory of population size of the model fit to 1980-94 to estimates of population size derived directly from 1995-99 data.

Their analysis compared ‘observed’ population size including data through 1999, and the trajectory predicted by including data only through 1994 for several Snake River spring/summer chinook index stocks. An example is shown for Poverty Flat (Figure 3). The CRI method fits the time series in essence by passing a line through two points: the total population estimate at the beginning of the period (1984 point) to the estimate at the end of the period (1999). When USFWS re-estimated ë, using data only through 1994, a much higher trajectory resulted, because the 1994 population estimate is much higher than the 1999 estimate. Thus the Dennis-Holmes model consistently over-predicts the status of the Snake River stocks when it is used to predict population numbers observed over the most recent years. Because of the variability in the data, it is unreasonable to expect the Dennis-Holmes model or any model to be able to predict future population trends exactly. However, a valid method would not systematically over-predict future trajectories among different stocks; the expected trajectories should be equally likely to be optimistic or pessimistic, compared to the actual outcome. The fact that applying the Dennis-Holmes model to 1980-94 data seems to miss the recent downward trends in population abundance, and consistently and drastically overestimates the actual recent population trends, is troubling. It suggests decision makers can have little confidence in near term projections of population growth or decline (driven by 8) arising from the Dennis-Holmes model. Further, this problem translates into a consistent under-prediction of the probability of extinction for these populations.

Corroboration with different time periods

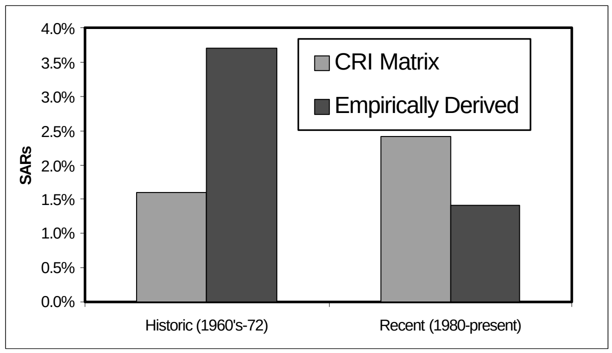
In an attempt to validate the CRI matrix model’s ability to accurately predict the status of the population and the distribution of mortality across the life cycle, STUFA used the CRI matrix to estimate historic (1957-1967) smolt-to-adult survival rates (SAR) values from the time period before most of the lower Snake River dams were present. SARs include the time period after the smolts leave the hydrosystem below Bonneville to the time they return to Bonneville dam as adults (Figure 1). This is the time period for which PATH links delayed mortality, in part, to hydrosystem experience. PATH life cycle models directly incorporate results from retrospective analyses about the past into prospective modeling of management actions such that the observed patterns of salmon decline, and SAR's, are automatically included in analyses. Since the CRI matrix model does not consider past data or patterns, this exercise was meant to answer the question, "how well does the CRI matrix model account for the different sources of mortality across the life cycle if used to predict population growth rates and survival measures for a different time period?".

Figure 3. Comparison of the method's predicted escapements to the observed data for Poverty Flat, one of the PATH index stocks, from the NMFS Dennis-Holmes model. Shown are comparisons of 1995-99 predictions using the a 1980-94 model fit to observed data. The 'running sum' estimate is an estimate of total population size; that is, not only spawners but fish alive of any age. The triangles that fall after 1994 represent the prediction of the Dennis-Holmes model for 1995-99, and the squares represent the observed escapement.



The CRI model was used to try and capture past patterns during the time period when populations were relatively healthy and these stocks migrated through only three to five dams rather than eight dams. Harvest rates, adult passage, and direct downstream survival rates through the hydrosystem were adjusted to reflect the earlier time period. For this time period, CRI predicts that SAR values were substantially lower than present SAR values (Figure 4). In other words, the CRI model predicts that smolt-to-adult survival is much higher now than in the past, before the hydrosystem impacted these stocks, a result that is not supported by the data.

Figure 4. SAR's (smolt-to-adult survival) for spring/summer chinook, from Lower Granite dam as smolts to the time they return to Bonneville dam as adults (*see Figure 1*). Lighter bars show values predicted by the NMFS CRI matrix model. Darker bars represent actual values estimated from PIT tag data. The graph demonstrates that the NMFS CRI matrix model can not predict historical trends in smolt-to-adult survival and instead predicts the opposite pattern to what was observed. These results indicate that the NMFS matrix model is not accounting for some additional source of mortality (e.g. extra or delayed mortality) that occurred after the completion of the hydrosystem.



The inability of the CRI model to predict the decline in smolt-to-adult survival that has occurred since the 60's indicates that the model is <u>not</u> accounting for an important source of mortality in some life stage. It appears likely that the mortality that is <u>not</u> being accounted for occurs in the stages after juveniles have migrated to below the hydrosystem since differences in passage survival and harvest rates were incorporated in the analysis. This exercise, in addition to others, indicate that the NMFS matrix model may underestimate the effectiveness of dam breach because delayed mortality due to stresses associated with the hydrosystem is not explicitly included in their analyses, and because they do not incorporate past population declines.

Modeling management actions

The STUFA analysis also evaluated the results of directly addressing the impact of alternative management actions with the CRI matrix model. In the original CRI model, NMFS evaluated potential management actions by hypothetically asking "what would be the benefit to the population by decreasing mortality 10% in each life stage". However, these "hypothetical numerical experiments" do not necessarily correspond to biologically and logistically feasible actions aimed at improving the health of these stocks. For example, although a hypothetical 10% reduction in egg-to-smolt mortality may be sufficient to halt the decline of these stocks, a 10% reduction in mortality at that life stage may not even be possible (e.g. some stocks spawn and rear in wilderness areas where little can be done to further reduce egg-to-smolt survival). Further, using this approach is likely to always demonstrate that life stages with the lowest assumed survival will provide the largest benefit to the population, but that increases in survival may not be biologically possible within the evolutionary constraints to the life history of this species (e.g. chinook produce up to 6000 eggs per female and may naturally have a high mortality rate in this life stage).

Based on these concerns, STUFA explicitly modeled feasible alternative management actions using a modified CRI matrix type model. The management actions that were modeled were chosen because they were similar to some of the management actions modeled in PATH and allowed for a general comparison between the results of the two analytical approaches. Management actions modeled using the modified CRI model included:

1) *Baseline*: "Average", status quo conditions from 1980 to present (similar to the *A1* scenario in PATH),
2) *Maximize transport*: Maximizing the number of smolts transported at each collector project (similar to the *A2* scenario in PATH),
3) *Dam breach/delayed mortality reduced*: Four lower Snake River dams are breached and delayed mortality is assumed to be linked to the hydrosystem and is thus reduced (similar to the *A3/Hydro* scenario in PATH),
4) *Dam breach/delayed mortality remains*: Four lower Snake River dams are breached, but delayed mortality of non-transported fish is assumed to be unrelated to the hydrosystem and remains (similar to the *A3/BKD* scenario in PATH),
5) *Everything but dam breach*: All feasible management actions except dam breach. Maximize transport plus zero harvest, improvements in habitat, and reduction in estuary smolt mortality via Caspian tern relocation, and

6) *Everything including dam breach*: All feasible management actions including dam breach. Includes zero harvest, improvements in habitat, and reduction in smolt mortality via Caspian tern relocation, and dam breach (with delayed mortality reduced).

The modified CRI matrix model indicated that little improvement in population growth rates (lambda) or changes in smolt-to-adult survival rates (SARs) over current conditions (*Baseline*) would be expected under the *Maximize transport* management scenario. The greatest improvement occurred under the dam breach scenarios where average annual population growth rates were positive and increased by approximately 30% and SARs increase 205 to 235%. Improvements were not as great under dam breach if transport delayed mortality was assumed low (‘*D’* assumed high) and extra mortality of non-transported fish remained after dam breach; however, improvements were still higher than *Maximize transport* scenarios. Thus in general, when the CRI matrix type model was used to explicitly model management actions with feasibility considered, the results corresponded closely to those from PATH analyses.

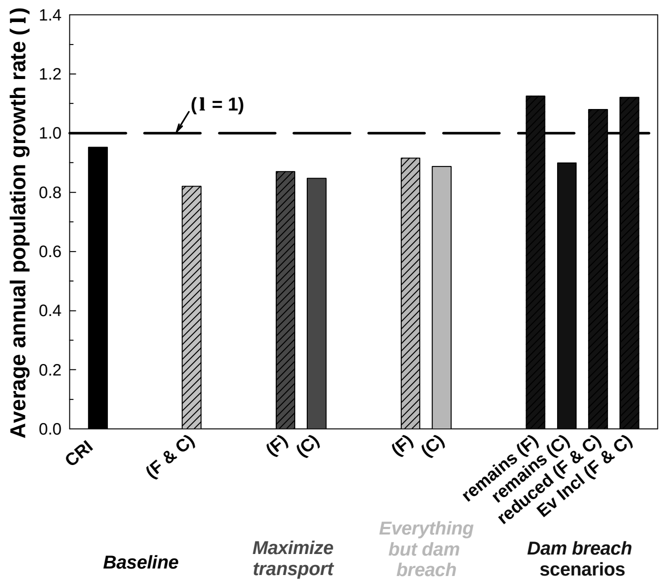
Accounting for variability and uncertainty

In addition to explicitly modeling feasible management actions, the STUFA analysis evaluated the effects of NMFS deterministic model, which essentially assumes everything in nature is stable and there is no measurement error or random climate or environmental effects on survival. In general, demographic models that incorporate variability around model parameters (e.g. PATH) tend to produce more pessimistic (conservative) results about population growth rates than models that assume parameters are fixed. Therefore, STUFA incorporated a distribution of potential values for each model parameter, a way of including variability or stochasticity, into the CRI matrix model. As expected, the median annual population growth rates under all scenarios evaluated by the modified stochastic matrix were lower than the population growth rates determined from the CRI matrix model for any given scenario.

For the different management actions, when variability was included, the *Maximize transport* scenario did not provide much benefit over *Baseline,* with the probability of a positive population growth rate of just 25% (Figure 5). The *Everything but dam breach* scenario yielded little improvement over the *Maximize transport* scenario alone. The *Dam breach* scenarios had the greatest benefit to the stocks with three of the four scenarios having a 65 to 75% probability of a positive population growth rate. The greatest improvement occurred when *Everything including dam breach* was implemented; however, the population growth rates only increased approximately <u>4% over the dam breach alone</u> scenario. Again, these results were consistent in pattern to the results from the PATH and the deterministic CRI model (above) that was modified to model feasible management actions. But as expected, all population growth rates were <u>lower</u> for any given action, because of the incorporation of uncertainty or variability. In later analyses, NMFS adjusted the amount of survival needed to halt the decline in these stocks upward to reflect the lack of variability or stochasticity. Nevertheless, the STUFA analysis demonstrates that regardless of what action is being considered, incorporating variability produces lower growth rates and indicates that deterministic CRI results may have been optimistic and thus risky.

The STUFA analysis demonstrated that the CRI conclusion that improvements in transportation and the hydrosystem short of breaching will not provide much benefit to these stocks is consistent with PATH. In addition, the STUFA analysis also showed that both PATH and CRI are consistent in demonstrating that a reduction in harvest will not provide much benefit to spring/summer chinook, largely because harvest is already minimal.

Figure 5. Results from the STUFA evaluation of the NMFS CRI model and comparison to PATH for the NMFS matrix model modified to explicitly model management actions and feasibility within a stochastic (random variability) structure. Lambda, or the average annual population growth rate, is shown on the y axis with the different groups of management actions shown on the x axis. F stands for the FLUSH passage model and C stands for the CRiSP passage model. Dam breach options have the highest population growth rates compared to the other management actions.



Conclusions from the STUFA evaluation

Dam Breaching:

Despite the apparent differences between PATH and CRI, the STUFA analysis demonstrated that the CRI model results are generally similar to those from PATH. Instead, it is the underlying assumptions that drive differences between the two approaches. Results from NMFS original numerical experiments suggest that the management action that would likely provide the greatest benefit to these stocks is to implement <u>all</u> proposed management actions <u>including</u> dam breach (NMFS, 1999). The STUFA analysis and PATH analyses led to the same conclusion; however, in the STUFA review, the improvements in survival from everything <u>including</u> dam breach <u>over</u> those from dam breach <u>alone</u> were not as great. In addition, PATH results indicate a greater benefit from dam breaching alone relative to the NMFS CRI models.

The greatest difference between the two approaches comes not from the model structure (which is very different), but from differences in the underlying assumptions about the effect of the hydrosystem on delayed mortality in the estuary and early ocean. By incorporating the delayed mortality component into the CRI matrix model, STUFA found that the relative rank of expected benefits from management actions evaluated in PATH and the modified CRI model were consistent. Dam breaching provided a much greater benefit to Snake River spring/summer chinook under a wider range of uncertainties than did the maximized transportation options. However, because CRI does not consider evidence that indicates that delayed hydrosystem caused mortality is substantial, they do not explicitly include this source of mortality in their model. Therefore, CRI has determined that dam breaching alone will result in a minimal benefit to these populations. See below for a discussion of evidence that indicates the hydrosystem is likely responsible for a large portion of the extra mortality that Snake River fish experience relative to lower river stocks. In addition, because the PATH analysis was a decision analysis that included multiple alternative hypotheses about delayed mortality, interpretation of PATH results is not hampered by this remaining uncertainty about delayed mortality. Under the PATH results, regardless of the cause and degree of extra mortality, and considering additional key uncertainties not discussed here, dam breach has the greatest probability of recovery and is the least risky.

The other substantial and unresolved difference between PATH and CRI lies in the CRI conclusion that the greatest benefits to these stocks would occur through improvements to spawning/rearing habitat, and estuary/early ocean habitat (independent of influences from juvenile hydrosystem migration). The benefits from spawning and rearing habitat improvements suggested by CRI are purely hypothetical, as demonstrated by the STUFA analysis. In the STUFA analysis, they restricted the amount of these improvements in spawning and rearing habitat survival by limiting stock productivity to the highest productivity estimate observed from all the stocks. They also restricted the benefit of improvements in the estuary (other than delayed mortality) to the improvement attainable from reduction in Caspian tern predation. Neither feasible improvements in egg-to-smolt survival nor a reduction of predators in the estuary provided a large or sufficient benefit to these stocks. See below for a discussion of the potential for increasing survival through improvements in the freshwater spawning and rearing stage. Differences in the CRI and STUFA model results highlight the importance of incorporating feasibility into modeled management scenarios. Without this crucial step, the STUFA analysis demonstrates that numerical experiments are misleading as they have the potential for highlighting the life stage with the lowest survival rate as key for management rather than the life stage that is actually <u>limiting</u> these stocks.

V. SIMPLE EXPLANATION OF DELAYED AND EXTRA MORTALITY.

In both the PATH and the CRI analyses, the efficacy of dam breaching to recover Snake River spring/summer chinook is largely dependent on whether, and the degree to which, delayed mortality in the estuary and early ocean is related to a salmon smolt's experience through the hydrosystem. The following section provides a simple explanation of delayed mortality and then discusses empirical data and other lines of evidence for delayed hydrosystem mortality.

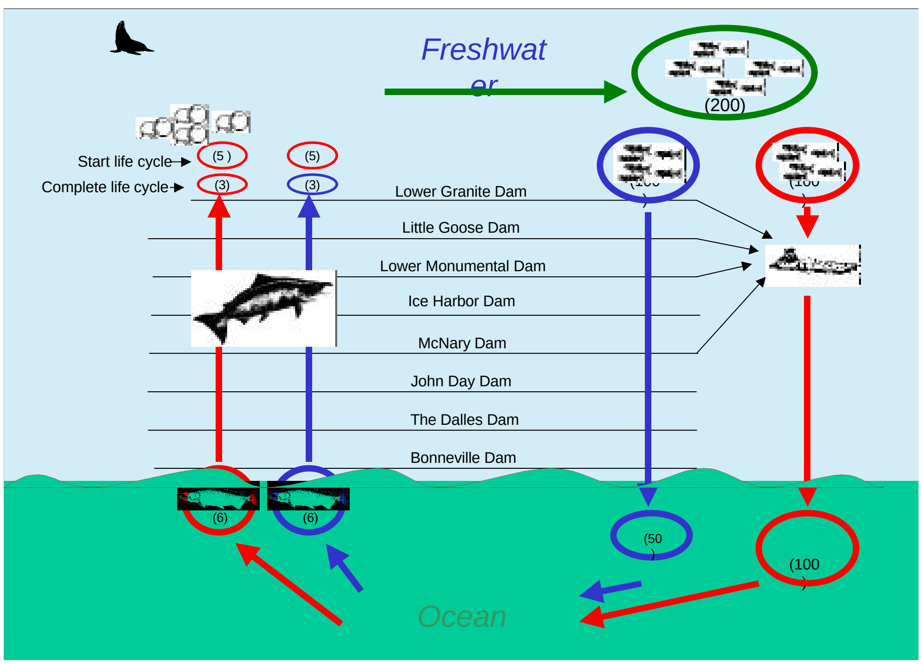
Non-technical example of delayed or extra mortality

For understanding limiting factors and modeling recovery options, salmon survival across the life cycle, from spawners to returning spawners (recruits) is often compartmentalized into survival at the various life stages (Figure 1): 1)egg to juvenile (rearing in freshwater), 2) survival during downstream migration (smolt stage), 3) survival in the estuary and early ocean, 4) survival during upstream migration as adults. For some of the life stages, there are data that allow estimation of the <u>direct</u> mortality that occurs in that life stage (how many fish die). Mortality that occurs in the <u>same</u> life stage as the <u>cause</u> of the mortality is called <u>direct mortality,</u> as it occurs immediately in that life stage (e.g., harvest or turbine mortality). What is less certain and more difficult to estimate, however, is how a fish's experience in one life stage may affect its survival in a later life stage. This mortality is called <u>delayed</u> (and extra mortality in some documents) and is similar to the case where a human who smoked cigarettes when they were younger, later dies of lung cancer. That person does not die at the moment they smoke their first cigarette, but they may die later as a result of the interaction between this earlier experience and long term health and fitness.

Fish originating in the Snake River must migrate past 8 hydroelectric dams on their way to and from the ocean. Not all fish can successfully migrate past the dams. Some are killed by turbines, and some are killed by predators in the reservoirs below the dams. The <u>direct</u> mortality that occurs when these juveniles migrate past each of the dams determines the survival in that life stage. The dams have sensors that can detect juveniles and allows the estimation of the survival from the top of the hydrosystem in the Snake River (from the uppermost dam, Lower Granite Dam) down to the lowermost dam in the Columbia River, near the estuary (Bonneville Dam), or over some stretch in-between. Juvenile salmon are also transported around the dams. The juveniles are generally collected at the three uppermost dams on the Snake River (Lower Granite) and transported by barge or truck down-river and released below Bonneville dam.

The direct mortality of fish that are transported around the dams (from collection at Lower Granite Dam to release below Bonneville Dam) appears to be quite low (close to 2%). This mortality of transported fish is considerably lower than the direct mortality of fish that travel in the river and have to migrate past the dams (~50-80%). However, as discussed above, this direct mortality is only one component of the overall survival to adult spawners. It is the overall survival to adults that describes the effectiveness of transportation and the effects of in-river passage past dams. The quantity which describes the <u>difference</u> between the delayed mortality of transported fish and fish that migrate in-river, is called ‘D’ (a ratio). ‘D’ would be equal to 1 if there were no difference between the two groups of fish. When the fish that traveled in-river survive better from the time they leave the hydrosystem to the time they return as adults, then ‘D’ is less than one (Figure 6).

Figure 6. Schematic diagram showing 'D'. The direct mortality of fish that are transported around the dams (from collection at Lower Granite Dam to release below Bonneville Dam) appears to be quite low (close to 2%). This mortality of transported fish is considerably lower than the direct mortality of fish that travel in the river and have to migrate past the dams (~50-80%). However, as discussed above, this direct mortality is only one component of the overall survival to adult spawners. It is the overall survival to adults that describes the effectiveness of transportation and the effects of in-river passage past dams. 'D' is a measure of the mortality of fish that were transported around the dams relative to fish that traveled in river, from the time they leave the hydrosystem to the time they return as adults (a ratio). D=1 indicates there is no difference; D<1 indicates the transported fish experience a higher rate of mortality after they leave the hydrosystem. The 'D' value describes, in part, the effectiveness of the transportation program (high D = more effective, low D= less effective).



For example, if the direct mortality of the transported fish were 0% and the direct mortality of the in-river fish were 50%, we would expect the transported fish to survive at 2 times the rate of in-river fish (because we lost 50% of the in-river fish in the hydrosystem). If the overall survival for the transport fish is 2 times that for in-river fish, there is no difference in the delayed mortality between the two groups (D=1). In this case, the fish survive at the same rate in the estuary and early ocean, regardless of whether they were transported or migrated in-river. However, for the same example, if the overall survival was the same for the transport and in-river groups (was not 2 times greater for transport fish), then the delayed mortality of transported fish must be twice as high compared to in-river migrating fish (again, because we lost 50% of the in-river fish in the hydrosystem). In this case, 'D'=0.5 and the survival of transported fish, from the time they leave the hydrosystem to the time they return as adults, is only half that of the fish that migrated in-river.

For both groups of fish, those that were transported and those that traveled in-river, there is evidence the delayed mortality that occurs in the estuary and early ocean is at least partially related to their experience either through the hydrosystem or during collection and transportation.Due to the stress of collection and bypass at the dams, and crowding during transportation in a barge or truck, transported fish may be more vulnerable to disease and predators later in life. Similarly, fish that travel in-river must successfully migrate past the turbines, bypass systems, and reservoir predators of eight hydroelectric dams. Stress or injury from this experience may also cause the fish to be vulnerable to disease and predation, either later down in the hydrosystem or while the fish are in the estuary and ocean. Some data indicate that the delayed or extra mortality of fish populations that migrate past 8 hydroelectric dams may be much greater than that of fish populations with similar characteristics from lower down in the system, where they migrate past only 4 hydroelectric dams. Because of this difference, scientists hypothesized that hydrosystem experience, whether in the barge or in-river, increases the delayed mortality of fish that occurs in the estuary or early ocean life stage. Specific evidence for delayed mortality is discussed in *Section VII below*.

VI. ASSESSING THE POTENTIAL FOR IMPROVING OVERALL SURVIVAL AND POPULATION GROWTH RATES THROUGH IMPROVEMENTS TO FRESHWATER SPAWNING AND REARING HABITAT.

As discussed in sections II, III and IV above, modeling results from PATH and NMFS CRI analyses conflict regarding the potential for halting the decline in spring/summer chinook based on improvements to freshwater spawning and rearing habitat. PATH analyses indicate that feasible improvements to freshwater spawning and rearing habitat are insufficient for meeting survival and recovery thresholds. PATH evaluated habitat improvements through a sensitivity analysis that incorporated both biological constraints or upper limits on improving survival at this life stage and also the probability of significantly improving the habitat. Under their analysis, stocks that spawn and rear in wilderness areas with little habitat destruction were deemed to have little potential for improving survival (Marmorek and Peters, 1998 a-b). Based on this approach, the PATH analysis demonstrated that feasible habitat improvements are insufficient for meeting survival and recovery goals, do not change the rank of management actions (dam breach still has the highest probability of recovery), and do not change the relative risk of the different actions (<u>dam breach is still the least risky option</u>).

In contrast, CRI analyses determined that the decline of these stocks could be halted if egg-smolt mortality was reduced by 3- 11%. Although they have not included practical or biological feasibility in these analyses, NMFS indicates that they believe that mortality could be reduced by the required amount based on hypothetical improvements to freshwater spawning and rearing habitat (NMFS 1999, NMFS 2000 a-d). Although the STUFA evaluation resolved many of the differences between PATH and CRI, this difference over the potential for improvement and halting the decline in spring/summer chinook was not resolved. The difference is due, in part, to how the different models allocate mortality and whether or not experience at one life stage is linked to mortality at later life stages (delayed mortality). However, the difference is also due to whether or not practical and biological feasibility is included in the analysis (PATH includes, CRI does not).

In this section, we discuss the pattern of <u>freshwater spawning and rearing survival</u> (Petrosky et al., *in press*) for spring/ summer chinook since the early 60's. We consider this analysis as it contributes to the overall question of whether we can halt the decline through improvements in survival of other life stages (i.e., freshwater spawning and rearing habitat), or whether further survival improvements are needed through Snake River dam breaching. Under this analysis, if freshwater spawning and rearing survival presents an opportunity for halting the decline and improving overall survival across the life cycle to levels that will lead to recovery, we would expect to see a decline in survival at this life stage over the past that explains a large portion of the overall decline. In other words, if habitat improvements are going to halt the decline, then habitat destruction and degradation must have caused a decline in freshwater spawning and rearing survival that can now be reversed.

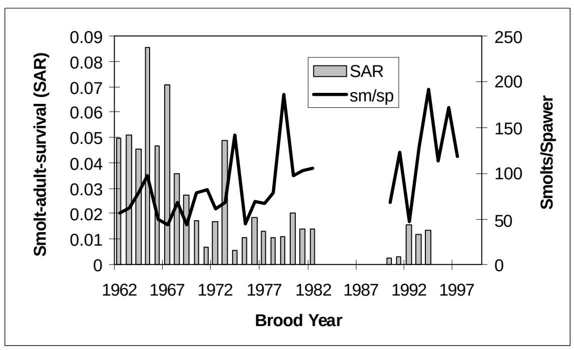
Approach- Smolts per Spawner

A more detailed description of this analysis is available in Petrosky et al. (*in press*). To evaluate trends in freshwater spawning and rearing (FSR) survival or productivity, Petrosky et al. *(in press*) developed a time series of spawner and smolt estimates at the uppermost dam (e.g, Lower Granite Dam) (Figure 1). The relationship between spawners (or parents) and smolts (or progeny) describes survival in the FSR stage, <u>after</u> spawning and <u>before</u> smolts migrate through the hydrosystem. They also considered the pattern of smolt-to-adult survival (SAR), which describes survival during later life stages, during downstream migration and in the estuary and ocean. In addition, their analysis considered how spawner numbers and smolt estimates were affected by uncertainties such as hatchery fish contribution and assumptions about guidance and flow at Lower Granite dam that may alter estimates of smolts. Their analysis evaluated temporal patterns and how these indices of survival and productivity changed after the completion of the hydrosystem.

Results- No decline in freshwater spawning and rearing survival

The Petrosky et al. (*in press*) analysis demonstrated several important results. As expected, individual estimates of spawners and of smolts at the uppermost dam declined through time and were significantly lower during the second time period, after the completion of the hydrosystem. However, estimates of FSR survival and productivity did not show this same pattern, and the number of smolts per spawner did not decline through time (Figure 7). Instead the estimates of the number of smolts/spawner showed a slight increasing trend through time. This slight increasing pattern is inconsistent with a theory of decreasing freshwater spawning and rearing survival. The increase is consistent with density dependent survival and productivity where freshwater survival increases at lower spawner densities. In contrast to the lack of decline in freshwater spawning and rearing survival, SAR's (smolt-to-adult survival during downstream migration and in the estuary and ocean) declined dramatically. SAR's were significantly lower in the later time period, after the completion of the hydrosystem.

Figure 7. Smolt-to-adult survival rates (SAR) and smolts/spawner for those years in the time series where data were available. The SAR describes survival during mainstem downstream migration back to adults whereas the number of smolts per spawner describes freshwater productivity in upstream spawning and rearing areas. Declines in survival in the freshwater life stage can not explain the observed trend in declining overall survival and declining smolt-to-adult survival.



Summary of freshwater spawning and rearing survival analysis

The Petrosky et al. analysis found no evidence for a decline in freshwater spawning and rearing survival of a magnitude that could explain the overall decline in survival of Snake River spring/ summer chinook. However, they did demonstrate that survival during the downstream smolt and estuary/ocean stage (SAR) has declined dramatically and by a degree substantial enough to explain the overall decline in Snake River spring/ summer chinook. The lack of decline in freshwater spawning and rearing survival over the time period when these stocks have plummeted suggests that improvements to freshwater habitat may yield little increase in survival overall. In addition, based on these data, it appears unlikely that these improvements to freshwater habitat would be sufficient to halt the decline of these stocks given that a large part of the overall decline appears to have been caused by mortality associated with passage through and cumulative or delayed effects of the hydrosystem. These results are consistent with PATH results on the lack of potential for increases in survival from habitat improvements and consistent with the observation that several of the index stocks considered spawn and rear in wilderness areas with good to pristine habitat.

VII. THE POTENTIAL FOR FRESHWATER HABITAT IMPROVEMENTS; AND EVALUATION OF A SET OF INDEX AREAS IN IDAHO AND OREGON

Introduction

Recent modeling efforts by the National Marine Fisheries Service (NMFS) suggest that a modest increase in freshwater and estuarine survival of juvenile Snake River spring/summer chinook salmon could facilitate recovery of the Evolutionarily Significant Unit (ESU), without breaching dams (Kareiva et al. 2000). To assess the feasibility of attaining the necessary improvements in survival within freshwater spawning and rearing habitat, changes in stock productivity resulting from habitat improvements must be evaluated. To do this a subset of stocks from the aggregate Snake River ESU are being considered (Table 4). These index stocks were chosen because they represent the broad range of conditions that exist in the Snake River Basin. They span a range of management history (Table 5), habitat quality (Table 6), and geology, and all index areas have a high level of data availability. A detailed summary (in annotated table form) of index stocks proposed for use in this evaluation follows.

Land Management

Current and historical land management activities in the selected watersheds have likely contributed to the range of chinook salmon habitat conditions observed. Information on land management occurring on federally owned lands in the proposed watersheds is described. Private land management activities are noted where that information was available. See Table 5 below for more detail on land management and Table 6 for details on habitat conditions.

Potential for habitat improvement

Opportunities for improving spring/summer chinook salmon freshwater spawning and rearing habitat are good for some of the proposed streams and limited for others. The primary habitat impacts related to land management that could be addressed include sedimentation, high summer temperatures, and direct mortality in unscreened irrigation diversions. Some sources of reduced survival can be changed over a relatively short time (e.g., screening diversions), whereas it may take others (e.g., excessive fine sediments, stream width to depth ratio) longer to respond to restoration activities. This assessment of potential for improved conditions does not include any socio-economic considerations of implementing restoration activities, it is merely a coarse evaluation of where potential exists among these index streams.

Table 4. Descriptive summary of proposed streams.

Stock	HUC Code	Ownership [a]	Mgt. Des.[b]	Run	Ecoregion [c]	Geology
Bear Valley/Elk	17060205	Federal	managed	Spring	Northern Rockies	Granitic
Marsh C.	17060205	Federal	NRA	Spring	Northern Rockies	Granitic
Sulphur C.	17060205	Federal	Wildernes	Spring	Northern Rockies	Granitic
Poverty Flat	17060208	Federal	managed	Summer	Northern Rockies	Granitic
Johnson C.	17060208	mixed	managed	Summer	Northern Rockies	Granitic
Secesh R.	17060208	mixed	managed	Summer	Northern Rockies	Granitic
Lemhi R.	17060204	private	managed	Summer	Snake River Basin	Volcanic
U. Grande Ronde R.	17060104	mixed	managed	Spring	Blue Mountains	Mixed[d]
Catherine C.	17060104	mixed	managed	Spring	Blue Mountains	Mixed[d]
Lostine R.	17060105	mixed	NWSR	Spring	Blue Mountains	Granitic

a. The majority of land in the upper Middle Fork Salmon River (including Bear Valley, Marsh, and Sulphur) is federally owned. The Poverty flat section of the South Fork Salmon River (SFSR) is mainly under federal ownership. Johnson Creek has several private inholdings in the lower watershed. The Lemhi River valley is largely under private ownership, though portions of tributary watersheds are administered by the Bureau of Land Management (BLM) and the U.S. Forest Service (USFS). Ownership composition for the upper Grande Ronde River Subbasin (all land upstream from where the Wallowa River enters the Grande Ronde, including Catherine Creek) is 0.41%-BLM, 0.33%-tribal lands, 0.19%-state, 45.8%-USFS, and 53.26%-private (ODEQ 2000). The upper portion of the Lostine River is under USFS ownership, though the lower index area is within private lands (G. Sausen, Wallowa-Whitman National Forest, personal communication).

b. Mgt. Des. = Management Designation; managed areas are those that have forest/rangeland management activities occurring within the watershed; wilderness areas are those protected under the Wilderness Act of 1964; NRA is National Recreation Area; NWSR is National Wild and Scenic River.

c. Northern Rocky Mountain Ecoregion (Omernik 1987) is characterized by high mountains and mixed-conifer forests. Portions of this ecoregion overlaying the Idaho Batholith have coarse-grained, highly erosive soils (Boise National Forest and Payette National Forest 1995). The Snake River Basin Ecoregion is characterized by benchlands and mountain ranges, a sagebrush steppe vegetation community, and aridisol soil types (Omernik 1987). The Blue Mountains Ecoregion is characterized by low to high open mountains, mixed-conifer forests, and mollisol and inceptisol soil types (Omernik 1987).

d. Columbia River Basalt is dominant rock type found in Upper Grande Ronde Subbasin (including Catherine Creek), though sedimentary and other igneous rocks are exposed in areas (Wallowa-Whitman National Forest 2000).

Table 5. Land management summary for proposed streams.

	Land Use Activities					
Stream Name	Logging	Mining	Roads	Irrigation	Grazing	Other
Bear Valley/Elk [a]	Y	Y	Y	N	Y	.
Marsh C. [b]	Y	Y?	Y	Y	Y	.
Sulphur C. [c]	N	N	N	N	N	.
Poverty Flat [d]	Y	Y	Y	Y	Y	.
Johnson C. [e]	Y	Y	Y	Y	Y	.
Secesh R. [f]	Y	Y	Y	Y	Y	subdivision
Lemhi R. [g]	N	Y	Y	Y	Y	.
U. Grande Ronde R. [h]	Y	Y	Y	Y	Y	urbanization
Catherine C. [i]	Y	N	Y	Y	Y	agriculture
Lostine R. [j]	Y	Y	Y	Y	Y	.

a. Timber harvest in Bear Valley Creek is limited to post-and-pole sales (Marmorek and Peters (eds.) 1998). The Bear Valley Mine produced tailings piles that have contributed substantial volumes of sediment to the creek after the stream meandered into them. Active restoration projects sponsored by the Shoshone-Bannock Tribe to deal with mine related sediment problems have been implemented (Marmorek and Peters (eds.) 1998). The drainage historically contained roads and still does (Marmorek and Peters (eds.) 1998). Forest Road 582 may be a potential source of sediment to Bear Valley Creek (K. Grover-Weir, Boise National Forest, personal communication). Grazing is believed to be the most degrading land use activity occurring in the Bear Valley Creek watershed (Marmorek and Peters (eds.) 1998). A Bureau of Fisheries survey of Bear Valley Creek reported that livestock were a problem as early as 1941 (McIntosh et al. 1995). Grazing rights in the Elk Creek Allotment were recently (2000) purchased by the Bonneville Power Administration (BPA) as part of the Fish and Wildlife Program (J. Kershner, USFS, personal communication). Grazing continues in the Deer Creek and Bear Valley Creek Allotments (K. Grover-Weir, Boise National Forest, personal communication).

b. Timber harvest and road construction have occurred within the Marsh Creek watershed, but to a lesser extent than in Bear Valley Creek (Marmorek and Peters (eds.) 1998). Livestock grazing has caused some localized reductions in streambank stability and increased inputs of fine sediments, though these areas have been actively restored (Marmorek and Peters (eds.) 1998). Cattle grazing has been excluded from most of the watershed since 1993 as per order of the Sawtooth National Recreation Area (Marmorek and Peters (eds.) 1998). Sheep are still grazed in portions of the watershed and cattle use one small allotment in the headwaters of the stream (T. Montoya, Salmon-Challis National Forest, personal communication). One screened irrigation diversion located on lower Knapp Creek serves as a barrier to upstream migrating spawners (T. Montoya, Salmon-Challis National Forest, personal communication). Marmorek and Peters (eds. 1998) suggest that there is little spawning habitat upstream from it.

c. Sulphur Creek is perhaps in the best condition of all of the proposed streams, as it is the least managed of all watersheds. Timber harvest and road construction have not occurred in the watershed historically (Marmorek and Peters (eds.) 1998). Grazing in the Sulphur Creek watershed is limited to a small fenced horse pasture and any "slop-over" grazing from other allotments (which IDFG personnel believe is limited) (Marmorek and Peters (eds.) 1998). Bureau of Fisheries personnel surveying the area in 1941 noted that "all in all this is one of the best salmon stream tributaries to the Middle Fork and although relatively small in size, it can care for several thousand spawning salmon and should be protected and kept open?" (McIntosh et al. 1995).

d. Substantial timber harvest has occurred in the SFSR watershed historically (esp. post-WWII; Platts et al. 1989), though less than two percent of the Poverty Flat subwatershed has been harvested since the 1950s (Boise National Forest and Payette National Forest 1995). Historically placer mining occurred in the Poverty Flat stretch of the SFSR, but it has since been allowed to recover (Boise National Forest and Payette National Forest 1995). The Poverty Flat subwatershed historically had high a high road density, mainly for timber harvest activities (Platts et al. 1989). These roads contributed large volumes of sediment to the system during the 1964-65 winter rains. The South Fork Road was believed to be the source of about 30% of management related sediment in recent years (Platts et al. 1989), though it has been paved within the last five years, potentially reducing its impact (R. Nelson, Payette National Forest, personal communication). There are many road crossings within Riparian Habitat Conservation Areas (RHCAs) in the Poverty Flat area (Boise National Forest and Payette National Forest 2000). Water withdrawal in the SFSR is limited to USFS water rights for recreational and domestic purposes (Boise National Forest and Payette National Forest 1995). Portions of the SFSR watershed were severely overgrazed historically (seventy to ninety years ago), with effects seen still seen in altered overland flow (Boise National Forest and Payette National Forest 2000).

e. Timber harvest trends for the Johnson Creek watershed follow those of Poverty Flat/SFSR. Less than one percent of the Johnson Creek watershed has been logged since the 1950s (Boise National Forest and Payette National Forest 1995). Current harvest is limited to a small number of salvage sales and post-and-pole sales (J. Fischer, Boise National Forest, personal communication). Mining has occurred historically and currently occurs in lower Johnson Creek (Boise and Payette National Forests 1995, 2000). Limited active surface and subsurface mining occurs at a claim in the Golden Gate area of lower Johnson Creek (Boise National Forest and Payette National Forest 2000). Some water withdrawal occurs in lower Johnson Creek for private and USFS use (Boise National Forest and Payette National Forest 1995). Most of these diversions are screened or are scheduled to be screened (J. Fischer, Boise National Forest, personal communication). Boise National Forest and Payette National Forest (1995) report that roads may be the dominant source of management related fine sediment in lower Johnson Creek. Past livestock grazing may still influence the hydrology of Johnson Creek by altering the patterns of overland flow (Boise National Forest and Payette National Forest 1995). Currently grazing is limited to two allotments in the Johnson Creek watershed. One of these is voluntarily being rested, and the other is an active allotment utilized by fifteen horses (Boise National Forest and Payette National Forest 2000).

f. The management history of the Secesh River is similar to that reported for SFSR, though it has not experienced as much degradation due to a less steep terrain. The conditions in the Secesh River are among the best for chinook salmon in the SFSR subbasin (R. Nelson, Payette National Forest, personal communication). Residential development in the Secesh River watershed is greater than in any other portion of the SFSR subbasin; however, with a subdivision in development approximately 25 km upstream of where the Secesh River enters the SFSR.

g. The most influential land management activity that appears to be occurring is water withdrawal for agriculture/livestock grazing (*see*

Dorratcaque 1986 and DerHovanisian and Megargle 1998). There are a number of diversions throughout the watershed, most of which have been screened and studied extensively. Agricultural and residential development in the Lemhi River watershed is extensive, and much of the land is in private ownership.

h. Timber harvest in the upper Grande Ronde River watershed has occurred since the late 1800s and has been steadily increasing since the 1950s (McIntosh et al. 1994a; McIntosh et al. 1994b), though harvest has slowed substantially in the 1990s (Wallowa-Whitman National Forest 2000). Splash dams were often used to transport timber via waterways, and have been noted as a habitat degrading remnant of historical timber harvest activities (McIntosh et al. 1994a; McIntosh et al. 1994b). Also, railroads constructed for transporting timber out of the uplands has constrained reaches of the Grande Ronde River (Wallowa-Whitman National Forest 2000). Mining activities in the watershed have contributed to degraded chinook habitat conditions as well. Tailings piles have constrained the channel in some areas and serve as chronic sources of sediment (affecting nearly 5 km of stream), many located in important chinook spawning locations (McIntosh et al. 1994a; McIntosh et al. 1994b; Wallowa-Whitman National Forest 2000). A portion of these mining sites are designated historical monuments and can not be actively restored as a result. Road density in the index portion of the upper Grande Ronde watershed in approximately 2.1 km of road per square km. Irrigation diversions do not exist in the index reach of the upper Grande Ronde, though there are numerous diversions downstream (Wallowa-Whitman National Forest 2000). Portions of this watershed were severely overgrazed as early as the 1880s but conditions have since improved substantially. Grazing continues to occur, though riparian fences exist in some areas and a variety of rotation schemes are being employed to minimize negative impact (Wallowa-Whitman National Forest 2000). Urbanization is substantial in the lower portion of the upper Grande Ronde River Subbasin (downstream of the index area).

i. Land management trends in the Catherine Creek watershed have mirrored those of the Upper Grande Ronde watershed as a whole. Timber Production has increased since the 1950s (McIntosh et al. 1994a, 1994b). There is no history of mining activity in the Catherine Creek drainage (Wallowa-Whitman National Forest 2000). Current density in the Catherine Creek watershed is 1.5 km road per square km (Wallowa-Whitman National Forest 2000). Many kilometers of road have been obliterated in the drainage and more are scheduled for obliteration. Irrigation diversions in Catherine Creek served as migration barriers as recent as the 1960s (Marmorek and Peters (eds.) 1998). Withdrawals of water on Catherine Creek have been described as having a low to moderate effect on chinook survival (Mobrand and Lestelle 1997). Approximately 48 km of lower Catherine Creek are rendered unusable at times by spawning and rearing chinook due to high summer temperatures (possibly related to water withdrawl) (Wallowa-Whitman National Forest 2000). Grazing has contributed to substantial stream bank loss in unexclosed portions of the Catherine Creek watershed (Kaufmann et al. 1983).

j. Timber harvest in the Lostine River watershed occurs mostly on private lands in the lower portion of the watershed (G. Sausen, Wallowa-Whitman National Forest, personal communication). There are currently no mining activities on federal lands within the watershed. Few problem roads exist in the watershed, though some feel that a USFS road running adjacent to the stream for approximately 50 km is a potential source of sediment. Numerous irrigation diversions are located on private lands in the lower Lostine River (Marmorek and Peters (eds.) 1998). Livestock grazing in the Lostine River drainage occurs primarily on privately owned lands (G. Sausen, Wallowa-Whitman National Forest, personal communication).

The upper Middle Fork Salmon River index streams (Bear Valley/Elk, Marsh, and Sulphur index stocks) have mixed potential for improvement. In Sulphur Creek there is little that will improve existing habitat conditions for this stock, as it is likely at its maximum habitat potential already, and its protective status (within Frank Church River of No Return Wilderness) will likely prevent any future impacts to its watershed. Opportunities in Marsh Creek appear to be quite limited as well, as land management related impacts are few. Impacts from past livestock grazing activities (localized sedimentation, reduced streambank stability in areas; Marmorek and Peters (eds.) 1998) are still visible in the watershed. It may take over twenty years for channel properties to return to natural levels after livestock are removed from a stream (Kondolf 1993), so it is likely that Marsh will continue to improve (grazing was ended in 1993; Marmorek and Peters (eds.) 1998). Of all Middle Fork Salmon River index stocks, the Bear Valley/Elk stock has the most potential for improvement. Opportunities for reductions in sediment input likely lie with livestock management. The elimination/alteration of grazing activities could substantially reduce its influence on fish habitat, though the benefits of livestock removal or riparian fencing can take decades to be realized (Kondolf 1993). Other opportunities for reducing inputs lie in the continued restoration of the Bear Valley Mine site and in surfacing or obliterating influential roads. Elimination or modification of harmful activities within the Bear Valley Creek watershed coupled with active restoration of sediment sources (e.g., revegetation of destabilized stream banks) will likely result in improved conditions, though results will not likely be realized in the short term (5 years).

Table 6. Habitat and water quality conditions for proposed streams.

	Habitat Quality Rating [a]			Percent of stream length with rating [b]				sec. 303(d) listings [d]		
Stream Name	S/R	DR	OW	Excellent	Good	Fair	Poor [c]	Sed.	Temp.	other
Bear Valley/Elk	3	2	1	17	61	22	0	Y[e]	N	None
Marsh C.	2	1	1	41	30	20	9	N[f]	N	None
Sulphur C.	1	1	1	43	19	38	0	N	N	None
Poverty Flat	3	3	2	0	54	46	0	Y[g]	N	None
Johnson C.	2	2	2	7	67	26	0	Y[h]	N[h]	None
Secesh R.	.	.	.	24	76	0	0	N	N	None
Lemhi R.	.	.	.	16	33	48	3	Y	N	n,f
U. Grande Ronde R.	.	.	.	0	17	34	49	Y[i]	Y[i]	n,h,f,d
Catherine C.	.	.	.	0	0	59	41	Y[i]	Y[i]	n,h,f,d,p
Lostine R.	.	.	.	0	45	0	55	Y[j]	N	h,f

a. From Marmorek (ed., 1996). S/R = spawning and rearing; DR = downstream rearing; and OW = overwinter; 1 = high, 2 = intermediate, and 3 = low. These ratings were a result of a qualitative assessment performed by state agencies used primarily for ranking purposes and PATH modeling.

b. Data from NWPPC 1990/1991 subbasin planning, available on Streamnet. Habitat ratings (Excellent, Good, Fair, and Poor) were assigned to reaches defined by three categories of spring/summer chinook salmon use: migration, spawning and rearing, and rearing and migration. Data represented are only those defined as spawning and rearing and rearing and migration as reaches used primarily as migration corridors were not rated. Not all stream reaches were included in survey. All habitat ratings were assigned by professionals with local expertise on the given stream/watershed. Stream lengths included in calculation were all main-stem reaches and tributaries upstream from (and including) PATH index areas.

c. Reaches assessed on SFSR below the Poverty Flat index area were mainly rated poor for spawning and rearing. Most reaches on the Grande Ronde River downstream from PATH index areas (defined use: rearing and migration) were rated poor to fair.

d. Parameters for which the stream, or a given reach is identified as water quality limited under section 303(d) of the Clean Water Act. Only those that most affect fish or those affecting fish in their migrations are listed. Y = yes and N = no; n=excessive nutrients, h=habitat modification, f=flow alteration, d=dissolved oxygen, and p=pH. Sources: EPA's Surf Your Watershed, and ODEQ (2000).

e. Sedimentation has led to pool loss and degradation of spawning and rearing habitats in Bear Valley (Marmorek and Peters (eds.) 1998). Poor egg-to-parr survival or early downstream migration of juvenile chinook is a potential consequence of excessive fine sediments in Bear Valley (Scully and Petrosky 1991).

f. Local sedimentation may be a problem in Marsh Creek (Marmorek and Peters (eds.) 1998)

g. Sedimentation is largest factor contributing to degraded chinook salmon habitat in Poverty Flat (Boise National Forest and Payette National Forest 2000). Surface fine sediments are generally stable to increasing, and subsurface fines are decreasing (Boise National Forest and Payette National Forest 1995, 2000). Difficult to determine what the natural level of fine sediment in SFSR is, as a quantitative baseline of pre-1960s sediment characteristics is lacking (R. Nelson, Payette National Forest, personal communication).

h. Pools are limited in lower Johnson Creek, potentially due to sedimentation or lack of coarse woody debris (Boise National Forest and Payette National Forest 1995). Percent fines for Lower Johnson rated as "fair" to "poor" with an average of 27.2 percent. The worst areas are in C-channels, which is also where chinook spawn (Boise National Forest and Payette National Forest 1995). Boise National Forest and Payette National Forest (2000) suggest that temperatures harmful to chinook salmon can be reached in Johnson Creek, but it is difficult to say if it is as a result of any land management practices.

i. Principal land uses responsible for water quality problems in the upper Grande Ronde (including Catherine Creek) are: forest disturbances (both within and outside of riparian areas), agricultural riparian and upland disturbances, road construction, and urban/suburban development (ODEQ 2000). Substantial pool loss has occurred in the upper Grande Ronde River as a result of sedimentation (McIntosh et al. 1994a, 1994b). The quality of habitats for chinook salmon has been severely reduced (affecting survival at many life stages) due to increased temperature, increased sedimentation, changes in flow, riparian alteration, and bank destabilization (Mobrand and Lestelle 1997). Most of the 303(d) listings in the "other" category occur below PATH index areas, but still affect the stock.

j. Temperature, flow modification, and the reduction of habitat diversity are all believed to have reduced survival of chinook salmon in the Lostine River (Mobrand and Lestelle 1997).

As in the Middle Fork Salmon River, efforts for habitat improvement in the South Fork Salmon River could reduce the input of fine sediment. Past management activities have shifted towards sediment reduction as early as the 1960s with a moratorium on road building and timber harvest (Platts et al. 1989), and current land management and restoration activities continue towards that end (Boise National Forest and Payette National Forest 2000). The Poverty Flat index area is still believed to be recovering from the impacts of 1960s, but it is difficult to determine whether the stream has reached equilibrium with no good pre-1960s baseline. Therefore opportunities for improvements in the Poverty Flat area are limited, though projects proposed in the SFSR Subbasin Review (Boise National Forest and Payette National Forest 2000) could facilitate further reductions in fine sediment levels. None of these are likely to result in any changes in the short term, but will contribute to the long term recovery of the system. Opportunities for improvements in the lower Secesh River are expected to be limited as well. A list of proposed sediment reduction projects for the lower Secesh River is also available in the SFSR Subbasin Review. All proposed projects would likely result in improved fish habitat conditions, though changes in stock survival probably would not be dramatic, as sediment conditions in the Secesh River watershed are among the best in the SFSR Subbasin based on natural geology (R. Nelson, Payette National Forest, personal communication).

In the SFSR watershed the Johnson Creek index area may have the most potential for improvements, as it has high levels of fine sediments, some unscreened irrigation diversions (though any unscreened diversions are scheduled for screening), and possibly excessive summer temperatures. Sediment reduction projects proposed in the SFSR Subbasin Review will likely result in improved conditions if implemented. Obliteration or surfacing of problem roads additional to proposed projects will also help improve conditions, though none of these activities will produce immediate results. Screening of diversions will likely result in an immediate benefit to chinook salmon survival in lower Johnson Creek. Complete closure of diversions and the subsequent return of diverted flows may result in a reduction of the maximum summer temperature observed depending on the current level of withdrawl (though Boise National Forest and Payette National Forest (2000) suggest that high summer temperatures may be a natural phenomenon unrelated to management activities). Limited published information is available detailing the impact of the few diversions located on lower Johnson Creek, limiting the validity of any assessment of benefits to be gained from their screening or closure.

Opportunities for improving chinook survival in the Lemhi River watershed are limited, based on the high proportion of private land. However, the screening of any unscreened diversions will likely produce immediate benefits to survival. Also, the return of diverted flows to the river would likely result in a positive effect on the stock (lower summer temperatures and increased instream flows). Since detailed information on other activities occurring on privately owned lands (e.g., grazing) is limited, it is difficult to assess how much they are impacting fish survival and what measures can be taken to alter their influence.

The selected index stocks in eastern Oregon (upper Grande Ronde River, Catherine Creek, and Lostine River) have perhaps the most potential for survival improvements of all of the stocks proposed. Opportunities for sediment reduction in the upper Grande Ronde index area lie in further reductions in road density (via road obliteration), continued restoration of mine tailings piles, as well as fencing of additional riparian areas to exclude cattle. These projects, coupled with streambank stabilization projects currently under way (Wallowa-Whitman National Forest 2000) are likely to result in long term habitat improvements via sediment reduction. Additional opportunities for augmenting the freshwater survival of this stock lie in reducing mortality due to unscreened diversions downstream, as well as returning diverted flows to increase available rearing habitat and reduce summer temperatures (this is not an issue directly in the index area, but is a pervasive issue downstream where juveniles may rear). Opportunities in Catherine Creek and the Lostine River index areas are similar to those in the upper Grande Ronde River index area. Excessive summer temperatures in these streams (U. Grande Ronde and Catherine Creek) have resulted in 303(d) listings, and returning diverted flows may drastically alter this limitation in the short term. Also, increased riparian shading via revegetation could also help reduce summer temperatures, as there is limited buffer width on streams flowing across private lands (Wallowa-Whitman National Forest 2000). The benefits of this activity would be manifested slowly. The primary limitation to improving habitat conditions in these watersheds is the high proportion of private lands in the most impacted areas. The Grande Ronde River Model Watershed Project has facilitated the cooperation of private land owners with chinook salmon recovery goals, making land ownership less of an impediment as it might be in other watersheds.

The selected index streams cover a wide range of ownership, management history, geology, and habitat conditions. Consequently, opportunities for improving spring/summer chinook salmon freshwater spawning and rearing habitat are diverse as well. At a coarse scale of resolution it appears that eastern Oregon stocks have the greatest potential for both short and long term habitat improvements (upper Grande Ronde River, Catherine Creek, and the Lostine River having roughly similar potential), with Sulphur Creek in central Idaho having the least potential of all index streams. Of SFSR Subbasin streams, the lower Johnson index area may have the greatest potential for improvement, both in the short (e.g., screening diversions, returning flows) and long term (continued reductions of fine sediment levels). Potential for the Secesh and Poverty Flat index areas are comparable, with modest long term improvements (further sediment reductions) possible. In the upper Middle Fork Salmon River, the Bear Valley/Elk index area may have the most potential for habitat improvement, though any benefits are likely to be long term (e.g., sediment reductions and channel recovery from livestock may take decades to be realized). Opportunities in Marsh Creek are intermediate between those of Bear Valley/Elk and Sulphur Creek.

VIII. SUMMARY OF THE EVIDENCE THAT LINKS A FISH'S EXPERIENCE IN THE HYDROSYSTEM TO THEIR SURVIVAL IN LATER LIFE STAGES: DOES DELAYED MORTALITY EXIST?

Implications of delayed mortality

There are important implications of delayed or extra mortality when evaluating the outcome of management actions for spring/summer chinook.   Extra mortality and 'D' are not explicitly modeled in the CRI approach; however, the overall outcomes are generally similar to those that result from the PATH approach. If there is a significant delayed mortality of transported fish (low 'D') relative to in-river fish, then dam breach of Snake River dams has a high probability of recovering these stocks under both PATH and NMFS CRI models. If there is no or little delayed mortality of transported fish (high 'D'), and the extra mortality of in-river fish is related to their hydrosystem experience, then dam breach still has a high probability of recovering salmon. The CRI effort does not explicitly employ the concepts of D and extra mortality in their analytical framework, in part, because NMFS scientists estimate that 'D' is high. Instead, CRI points out that it appears that removal of four dams must decrease mortality below Bonneville ('D' and extra mortality in PATH) by 9% (depending on other assumptions about current conditions). NMFS defers the question of whether or not field data or other lines of evidence support the conclusion that such an improvement can be made.

In PATH, dam breach has a low probability of recovering these fish only if there is little difference between the survival of transported and in-river fish from the time they left the hydrosystem until they returned as adults (D close to 1) and the overall survival of both groups is unrelated to hydrosystem experience. Thus the assumed value of 'D' and whether or not the extra mortality of in-river fish is related to hydrosystem experience changes the overall effectiveness of dam breach as a management action for spring/summer chinook. However because PATH is a decision analysis that incorporates both hypotheses that link hydrosystem experience to delayed mortality and hypotheses that don't, the results represent the best management option in the face of this uncertainty. CRI does not make firm conclusions about whether delayed mortality would likely be reduced under a dam breach option. Instead they indicate that experiments which address this uncertainty are complex and unlikely (Kareiva et al., 2000). They also note that even if the Snake dams were removed, the fish would still incur delayed effects from the dams on the lower Columbia River, if such effects do in fact exist.

Direct and indirect evidence for hydrosystem delayed mortality

In the rest of this section, we discuss direct and indirect evidence for the link between hydrosystem experience and delayed mortality. A more detailed description of this topic is provided in Budy et al. (*in review)*. As fish migrate through a reservoir and past a dam there are three possible outcomes of this experience: 1) they migrate successfully past the dam and experience no negative impacts from passage, 2) they can die immediately from mechanisms discussed above, or 3) they can survive while sustaining significant amounts of stress (Figure 8). This set of outcomes is possible for each of the eight hydroelectric dams that a Snake River fish must negotiate. The combination of the effects of stress and ultimately this mortality is termed delayed mortality. Delayed mortality can occur both within the hydrosystem and in the estuary and ocean. For the hydrosystem, stresses due to dam passage, handling, transportation, sub-optimal environment, and predators may severely weaken juvenile fish. In addition, these elements may have cumulative effects on the fishes physiologically responses to stress. Stress can be difficult to measure given that the physiological effects of stress may or may not be readily observable and therefore may often be underestimated. Even when signs of stress due to hydrosystem passage are observable, however, linking the stress to mortality can be difficult and has likely led to an underestimate of the magnitude of delayed mortality. However, general evidence for sources and the impacts of stress provide a biological basis that hydrosystem delayed mortality exists.

Evidence for delayed hydrosystem mortality comes in several forms. In the fisheries literature, delayed or latent mortality and the mechanisms that are likely responsible for delayed mortality are often referenced but less often directly quantified. Comparisons of spawner and recruitment data between stocks with similar life history characteristics but different hydrosystem experiences provide indirect evidence regarding the factors and extent of delayed hydrosystem mortality. Similarly, temporal comparisons of stock performance before and after the development of the hydrosystem provide additional indirect evidence of delayed hydrosystem mortality. In addition, there is direct evidence of delayed hydrosystem mortality from recent passive induced transponder (PIT)-tag data documenting individual hydrosystem passage and survival histories. It is the combination, however, of information from different life stages that provides the ability to establish and estimate delayed hydrosystem mortality and to link this mortality to factors of the earlier hydrosystem experience. Although a general summary is provided here, Budy et al. (*in review*) provide a detailed synthesis of these pieces of evidence for delayed hydrosystem mortality.

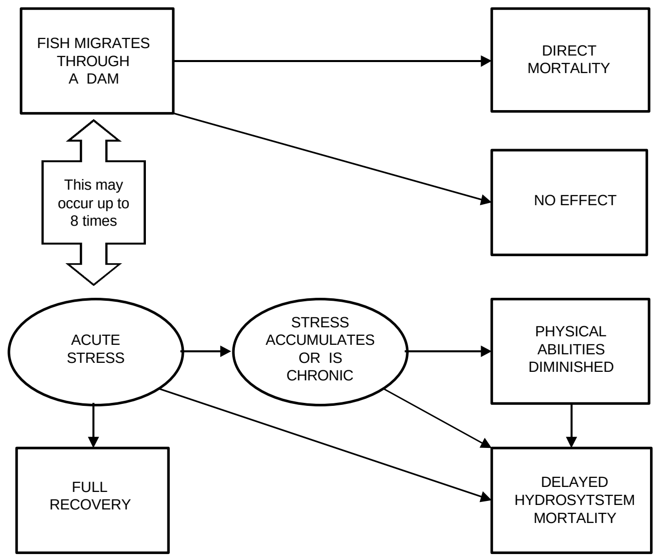
Evidence from the literature: a focus on impacts of stress

In Budy et al. (*in review*), they discuss and provide examples of stress events related to hydrosystem passage. Visible external signs of injury, trauma, and disease in fish have been recorded at dam collection facilities and are likely caused by dam passage and lengthened migration times through reservoirs (e.g., > 10% of the fish may be descaled; USACE 1998). However, these observations likely underestimate the true impact of each project because only visual signs were recorded. Similarly, there is evidence that a salmon's swimming performance, growth and blood chemistry are affected by exposure to high total dissolved gas levels, a response to stress that is not visible (Dawley and Ebel, 1975; Dawley et al., 1975; Wright and McLean, 1985; Krise et. al., 1990). Other stressful events from the hydrosystem likely manifest into similar diminishment of physiological response and performance. The hydrosystem has prolonged the migration time on Snake River salmon populations by decreasing water velocity and delaying fish at dams, and many investigators have concluded that salmonid smolts have "biological windows" to successfully enter the ocean (Walters et al., 1978; Bilton et al., 1982; Holtby et al., 1989 etc...). This extended freshwater residency is associated with reversion to parr-like characteristics with a lower salinity tolerance in some species of salmon (Conte and Wagner, 1965; Wagner, 1974 etc...), a potential source of delayed mortality in the estuary and early ocean stages.

In addition to stress events, Budy et al. *(in review*) discuss and provide examples of the decrease in a fish's energetic condition (e.g., energy reserves) in response to stress. In these cases, stressed fish would have less energy available for other activities (e.g., smoltification, disease resistance, swimming endurance, foraging ability, predator avoidance [e.g., burst speed, schooling]). Similarly, various types of environmental stress can also affect fish health and condition by lowering resistance to disease (*see* Wedemeyer, 1970), which can then lead to both direct and delayed mortality. The concept of increased vulnerability to predators as a result of stress is not new and is ubiquitous in ecology. Budy et al. (*in review*) provide numerous examples for fish, wolves, muskrat, birds etc… Due to the greater densities of piscivores and avian predators below Bonneville Dam (Ward et al., 1995; Roby et al., 1997), the interaction between predation and stress is a plausible mechanism causing hydrosystem delayed mortality.

Budy et al. also demonstrate that although stresses are generally measured as single point-specific events, stress may accumulate in fish and these additive effects may cause mortality at a later stage (Figure 8). Stress levels clearly increase as chinook smolts move through the collection and transport system at Lower Granite Dam (Mathews et al., 1986). There are numerous examples where the cumulative effects of multiple stresses on fish populations may cause the decline or collapse of a population, even though the effects of a single stress may be insignificant (*see review by* Wedemeyer et al., 1990). In addition, while measuring the magnitude of stress experienced by a fish can be difficult in itself, quantifying the true impacts of stress is further confounded by difficulties in relating the level of mortality to the stress that brought it on. And even though stress may be observed in a fish, this stress may not be ecologically significant (Sprague, 1971). Indeed, full recovery can often occur in individuals that may demonstrate signs of stress (Mesa et al., 1994). Conversely, individuals that no longer show signs of stress through individual indicators may still die from the cumulative effects of stress. Thus although stress has been demonstrated to eventually lead to mortality, the total impact of stress has likely been underestimated because of both the difficulties in measuring cumulative stress effects and the complex interaction between one or more stressors and multiple interactions with other sources of mortality.

Figure 8. Flow diagram of mortality and survival options as a juvenile fish migrates past a dam, up to eight times. As a fish passes a dam, it either dies, may experience no effect, or it may encounter stress. A fish may fully recover from these stresses or the stress may accumulate or have chronic health effects on the fish, leading to decreased physical abilities. Acute stress can lead directly to delayed mortality, or diminished abilities may lead to death via mechanisms of delayed mortality.



Indirect evidence: retrospective analyses and downstream stock comparisons

Retrospective comparisons of stock performance among regions and time periods provide indirect evidence that delayed mortality is linked to hydrosystem experience (Marmorek et al., 1996). These comparisons are based on spawner and recruit (offspring of spawners) data describing the overall survival (or mortality) over the life cycle of spring chinook salmon for brood years 1957-1990 (Figure 1). Delayed hydrosystem mortality, occurs in the Columbia River downstream from Bonneville Dam, in the estuary, and in the ocean. After accounting for direct hydrosystem mortality, natural spawner and recruit processes (e.g., density dependent mortality), and environmental or ocean effects, the delayed mortality the Snake River fish experience appears to range from about 37 to 68% (Marmorek et al., 1996; Marmorek and Peters, 1998 a-b). Two types of comparisons yield information regarding the degree or source of delayed mortality for Snake River stocks of spring and summer chinook. There are two important comparisons of overall survival rates for Snake River stocks that contribute to the evidence for delayed hydrosystem mortality: 1) SAR's (smolt-to-adult survival rates) before and after the development of the hydrosystem discussed in section IV above, and 2) stocks originating from different regions with different migration experiences and similar life histories discussed briefly in section II above.

Retrospective comparisons of stocks originating from different regions demonstrated that completion of the hydrosystem in the late 1960's through the mid-1970's was immediately followed by considerably sharper declines in SAR's and overall survival rates of Snake River stocks. The decline in SAR's was significantly greater after the completion of the hydrosystem and could not be explained solely by increased direct hydrosystem mortality (*see* sections II and IV above). Similarly, the decline in <u>overall</u> survival rates was greater in Snake River stocks than for downstream stocks (Schaller et al., 1999) over the same time period. The retrospective analysis was based on comparisons of survival rate patterns for Snake River (upstream) stocks to their counterparts downstream (which migrate through fewer dams). Therefore, after accounting for the difference in direct mortality between the upstream and downstream stocks, there was still an unexplained portion of mortality between the two stock groups, which accounts for the differences in overall survival patterns (*see* Marmorek and Peters, 1998 a-b). This unexplained mortality, that occurs in the estuary and early ocean and is specific to Snake River stocks, was estimated as a residual of the stock recruitment function after accounting for direct hydrosystem mortality and for all other known sources of mortality (e.g., density dependence, harvest, climate effects). The sharper declines in survival rates, for Snake River stocks, could not be explained entirely by the patterns of harvest, habitat degradation, climate, or hatcheries (*see* Sections II above and Schaller et al., 1999).

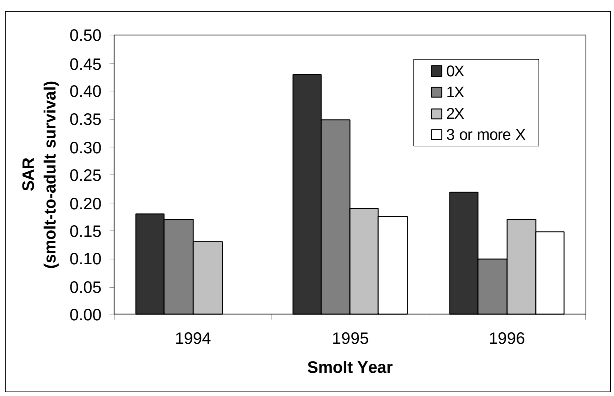
Direct evidence: relation to hydrosystem experience

Comparisons of overall survival of different groups of fish that experience different combinations of passage routes and transportation around the dams provide direct evidence of delayed hydrosystem mortality in relation to hydrosystem experience. In 1994-1996, NMFS expanded their studies on transportation and in-river survival of Snake River spring and summer chinook using PIT-tagged fish. These PIT-tag data record the history of a fish as it passes any place where a PIT-tag detector is located on a dam. When the juvenile fish migrate downriver, the PIT-tag detectors record their route of passage around each dam (bypass or transport); the fish are not detected if they pass by turbines or spill. When the adults return the detectors record their successful passage past the uppermost dam on the Snake River, Lower Granite Dam (Figure 1). This detection record allows the estimation of the overall survival rate (smolt-to-adult survival rate, SAR's) of various groups of fish that experience different passage routes throughout the Snake and Columbia Rivers. These data are discussed in greater detail in Bouwes et al. (1999), NMFS (1999), Sandford and Smith (1999), and Kiefer et al. (*in review*).

For fish that are not transported and migrate in-river, evidence for delayed hydrosystem mortality in relation to hydrosystem experience can be evaluated by comparing the overall SARs of fish that were not collected and bypassed around the dams with those fish that were collected/bypassed one or more times. Spill over the top of a dam is the route of passage most similar to a natural river, whereas collection and bypass has been shown to have a much higher potential of stressing the fish (discussed above). However, direct mortality is generally estimated to be similar between the spill and bypass routes, and lower than the turbine route (Marmorek and Peters, 1998 a-b). We would expect direct survival rates for collected and bypassed fish to be generally higher, because a portion of the fish that are not collected go through the turbines. Therefore, the more times fish go through the bypass system, the lower their direct mortality due to the hydrosystem and thus a higher smolt-to-adult return rate is expected (in the absence of delayed mortality).

The apparent, direct survival benefits of the bypass route of passage, however, do not translate well into SARs of wild spring and summer chinook. Estimates of SARs for in-river migrating smolts were examined each year for groups that experienced collection/bypass 0-, 1-, 2-, and 3 or more times through the four collector dams (Bouwes et al., 1999). These PIT-tag data indicate that SARs decreased when the number of times fish were bypassed increased (Figure 9; NMFS, 1999). Point estimates of SARs were consistently higher each year for those fish that were not bypassed at any of the four collector projects (zero times, 0X). Point estimates of SAR were intermediate for smolts bypassed one (1X) and two times (2X). Few to no adults returned from fish collected and bypassed three or more times (3 or more X), although sample sizes were small for this group in 1994 and 1996. These data further indicate that although <u>direct</u> mortality may be lowest for fish that are bypassed, there must be some <u>delayed</u> effect to explain the patterns of overall survival. Therefore these data provide empirical evidence that delayed mortality is related to hydrosystem experience.

Figure 9 . Estimated smolt-to-adult return rates (SAR) of wild spring and summer chinook by in-river passage history: non-detected (0X), and bypassed one, two, and three or more times (1X, 2X, 3 or more X, respectively), 1994-1996. These different groups of fish demonstrate increasing levels of delayed mortality based on their hydrosystem experience or route of passage.



In addition to comparing the overall survival of fish that migrate via different routes in-river, the smolt-to-adult survival rate of transported fish, relative to in-river fish (‘D’) provides evidence of hydrosystem delayed mortality. The direct mortality of fish that are transported through the hydrosystem appears to be very low (~2%), whereas the direct mortality of the fish that migrate in-river is generally greater than 50%. Therefore, if there were no additional delayed mortality that resulted from being collected and transported, we would expect the transported fish to return as adults at a rate nearly two times that of the in-river fish (given almost all transported fish survived through the hydrosystem and half of the in-river migrants died while migrating). After accounting for this estimated direct mortality, a post-hydrosystem SAR can be estimated. The ratio of transport to in-river post-hydrosystem SARs is referred to as 'D' (*see* section V. above). In the case of equal post-hydrosystem SARs of transported and in-river fish, 'D' would be equal to one. If this ratio is less than one, transported fish die at a higher rate in the estuary and ocean, after they leave the barges and trucks downstream of the hydrosystem, relative to in-river migrants.

The degree of the delayed mortality of transported fish relative to fish that migrate in-river, or 'D', has been a subject of debate. For Snake River spring and summer chinook, PATH developed models to estimate 'D' values for recent years, which averaged 0.48 using one model and 0.66 using another model. When estimates are made directly from the 1994-1996 PIT-tag data, estimates of 'D' for spring and summer chinook range from 0.59 (Bouwes et al., 1999 [1]) to 0.73 (NMFS, 2000). The 1994-98 average 'D' estimate is 0.52 using the revised Bouwes et al. method. The 'D' values by year using this method are: 1994 is 0.96; 1995 is 0.40; 1996 is 0.53; 1997 is 0.48; and 1998 is 0.41. NMFS used similar methods and assumptions to estimate 'D' for Snake River steelhead with values ranging from 0.52 to 0.58 (NMFS, 2000). PIT-tag estimates for fall chinook are not as reliable, but best estimates suggest that D=0.24 (Marmorek and Peters, 1998 b; NMFS 2000). While there is still some disagreement about the true 'D' values <u>all methods indicate that 'D' is considerably less than one</u>. This provides evidence that the delayed mortality exists, at least for transported fish. While 'D' simply describes the difference in

[1]Originally in Bouwes et al. the 1994-1996 average D value was equal to 0.49. However, the estimate was revised so that fish transported from different dams were weighted proportionally to the estimated proportion of nontagged fish transported from each dam so that the study fish were representative of the majority transported population (which was not tagged) resulting in a D=0.59.

delayed mortality between transported fish and fish that migrate in-river, it seems plausible that the absolute amount of delayed mortality for both groups may be substantially greater.

As discussed above, the hypothesis of delayed mortality due to hydrosystem passage has an empirical basis, as well as biological rationale. Based on recent PIT-tag data, there is direct evidence that delayed mortality of both in-river and transported smolts was related to the hydropower system. More specifically, the evidence suggests that, at least for collected and bypassed smolts, there is a difference between the patterns of direct passage survival rates and SARs. Smolts first detected and transported from the downstream projects (Lower Monumental and McNary) had lower SARs than smolts collected and transported from higher up in the system. Similarly, SARs of in-river smolts decreased as the number of times the fish were collected and bypassed increased (NMFS, 1999). These pieces of information provide <u>direct evidence</u> that the delayed mortality of Snake River spring and summer chinook is related to the juvenile migration hydrosystem experience.

Conclusion

Substantial evidence supports the existence of delayed mortality for Snake River spring and summer chinook and links delayed mortality to hydrosystem experience. This evidence comes in the form of published literature, indirect evidence from life-cycle modeling, and direct evidence from fish-tagging experiments. The literature supplies numerous mechanisms that explain how the observed stressors of the hydrosystem could be cumulative and eventually lead to mortality at a later life stage. The literature also demonstrates that the effects of stress can be hard to detect and difficult to empirically link to delayed mortality in field experiments. Nevertheless, there is little debate that fish condition and stress affect overall survival to adulthood or that the hydrosystem causes direct mortality and imposes stress to those fish that survive. Similarly, the retrospective life-cycle analysis provides indirect evidence that changes in delayed mortality have occurred in Snake River spring and summer chinook coincident with completion of the Snake River hydrosystem. The declines in survival rates of Snake River fish were considerably sharper than of downriver stocks over the same time period. Further, the survival rate declines occurred primarily in the smolt-to-adult life stage, rather than in the spawner-to-smolt life stage. PIT-tag data provides direct evidence that delayed mortality of both in-river migrating and transported smolts was related to the hydropower system. The patterns of smolt-to-adult survival rate could not be explained based on estimates of direct survival within the hydrosystem but instead must be explained by delayed mortality related to specific hydrosystem experience.

The compilation of evidence that links hydrosystem experience to delayed mortality has implications for the analyses of management options considered for the recovery of Snake River salmon and steelhead, especially the outcome of a dam breach option. In those analyses, dam breach is likely to dramatically increase the survival of Snake River chinook and lead to eventual recovery of these stocks if delayed mortality is related to hydrosystem experience. Dam breach is predicted to have less of an effect and may be insufficient for recovery if delayed mortality is unrelated to hydrosystem experience. Regardless of the action that is chosen to recover these stocks, the amount of hydrosystem mortality that needs to be compensated must include both direct and delayed components. Based on the evidence summarized in Budy et al. (*in review*), it seems implausible that little or none of the delayed mortality of Snake River fish is related to the hydrosystem.

IX. BOTTOM LINE

PATH

Prospective PATH analyses, or consideration of possible <u>future</u> management actions, for spring and summer and fall chinook salmon indicated that the dam breach actions had higher probabilities of achieving the survival and recovery standards than the “transportation” actions (Figure3). Dam breach actions met the standards over a wide range of assumptions (i.e. these actions are robust to remaining uncertainties). The dam breach actions met the 100-year survival and the 48-year recovery standards for spring/summer chinook even under the most pessimistic set of assumptions. The dam breach actions were also less risky than the "transportation" actions (i.e. model projections had relatively little variability over the full range of assumptions). For steelhead, a hydrosystem action was judged likely to meet survival and recovery standards if it meets the standards for Snake River spring and summer chinook salmon. Based on this comparison, the dam breach actions appear to have the highest probability of recovering steelhead.

PATH also performed detailed sensitivity analyses to a range of habitat improvements, additional predation (e.g., avian), and harvest restrictions. These analyses demonstrated that while it is likely that habitat improvements for some spring and summer chinook and steelhead stocks will increase the productive potential in the Snake River Basin, it does not appear that habitat improvement alone would be enough to recover these stocks. Additional predation did not change the rank of management actions as all actions are affected essentially equally –the dam breach actions still show the greatest probability of survival and recovery. Similarly, given the existing low levels of harvest on spring/summer chinook it is unlikely that additional future reductions in harvest alone will significantly increase the survival of spring/summer chinook such that jeopardy standards will be meet. Snake River fall chinook are harvested at a much higher rate in both the ocean (by both the U.S. and Canada) and in-river compared to spring and summer chinook. With the exception of one scenario, the dramatic harvest restrictions on fall chinook did not change the rank of management actions. However, if a 75% reduction in ocean harvest is combined with a 50% reduction in in-river harvest (that also requires recovery to be met before any non-subsistence level harvest can occur), then the transportation management action barely meets the recovery standard. It is important to note that these extremely dramatic changes would have to be in place forever. As soon as harvest restrictions were relaxed, the fall population would likely drop below recovery levels again.

NMFS CRI

For prospective analyses, NMFS evaluated the effect of eliminating all direct migration mortality except a small tribal subsistence harvest. This direct mortality includes only mortality that occurs immediately in the hydrosystem from turbines, bypass, and ladders etc... but <u>does not include</u> delayed or latent mortality that may be caused by the stress of hydrosystem passage or transportation but occurs later, in the estuary and ocean. This analysis showed that even with 0% mortality in river for juveniles and adults, stocks would continue to decline (KAREIVA et al., 2000). From these types of scenarios NMFS concluded that by itself, dam breaching is unlikely to recover spring/summer chinook salmon.

In addition, NMFS evaluated the effects of improvements in survival at other life stages (other than in-river migration) and whether these improvements would be enough to halt the decline in the <u>absence</u> of dam breach. They first concluded in the draft A-Fish Appendix (NMFS, 1999) that recovery of Snake River spring/summer chinook is likely to require dam breaching <u>and</u> many other aggressive management actions. Later analyses considered combinations of egg-to-smolt survival and early estuary and ocean survival that would halt the decline in the population (KAREIVA et al., 2000). These analyses showed that egg-to-smolt mortality must be reduced by 11% or early ocean /estuarine mortality by 9% to reduce the risk of extinction to acceptable levels. Delayed hydrosystem mortality may be expressed in the estuary stage.

For NMFS analyses, the effectiveness of dam breach hinges on the degree of delayed or latent mortality that is caused by the hydrosystem and experienced by spring/ summer chinook. Yet for some spring/summer chinook populations, NMFS indicates that extinction risks may be high enough such that to “delay action and study more” carries with it a substantial risk that some populations might be lost in the interim. Therefore, for the Snake spring/summer chinook, the <u>most risk averse action</u> would include dam breach, a harvest moratorium, and vigorous improvements in other areas as well, habitat, hatcheries etc...(NMFS, 1999).

Like PATH results, for Snake River fall chinook and steelhead, NMFS results are more clear and demonstrate that dam breaching by itself is likely to lead to recovery. However, temporary harvest reductions are also likely to reduce the short term risk of extinction for both fall chinook salmon and steelhead. NMFS notes however, that while dam breach will also increase available habitat for fall chinook (since they spawn and rear in the mainstem), harvest reductions will not affect the available habitat.

STUFA Analytical Comparison of CRI and PATH

The STUFA analysis demonstrated that the CRI conclusion that improvements in transportation and the hydrosystem short of breaching will not provide much benefit to these stocks is consistent with PATH. In addition, the STUFA analysis also showed that both PATH and CRI are consistent in demonstrating that a reduction in harvest will not provide much benefit to spring/summer chinook, largely because harvest is already minimal.

The greatest difference between the two approaches comes not from the model structure (which is very different), but from differences in the underlying assumptions about the effect of the hydrosystem on delayed mortality in the estuary and early ocean. By incorporating the delayed mortality component into the CRI matrix model, STUFA found that the relative rank of expected benefits from management actions evaluated in PATH and the modified CRI model were consistent. Dam breaching provided a much greater benefit to Snake River spring/summer chinook under a wider range of uncertainties than did the maximized transportation options.

Freshwater spawning and rearing survival.

Petrosky et al. evaluated trends in freshwater spawning and rearing survival based on the number of smolts per spawner from the 60's to the present (in press). Their analysis found no evidence for a decline in freshwater spawning and rearing survival of a magnitude that could explain the overall decline in survival of Snake River spring/ summer chinook. In contrast to the CRI results indicating a large potential for recovery based on freshwater habitat improvements, the lack of decline in freshwater spawning and rearing survival over the time period when these stocks have plummeted suggests that improvements to freshwater habitat may yield little increase in survival overall. In addition, based on these data, it appears unlikely that these improvements to freshwater habitat alone would be sufficient to halt the decline of these stocks given that a large part of the overall decline appears to have been caused by mortality associated with passage through and cumulative or delayed effects of the hydrosystem. These results are consistent with PATH results on the lack of potential for increases in survival from habitat improvements and consistent with the observation that several of the index stocks considered spawn and rear in wilderness areas with good to pristine habitat.

Freshwater habitat improvement potential; set of ten index areas

Based on an evaluation of ten index streams in Idaho and Oregon, it appears that while there is potential for improving habitat in some streams, in many of the streams, the degree of improvement that could be implemented and realized over the next 5-10 years is limited. For those streams where there is potential for improvement, the primary habitat impacts related to land management include sedimentation, high summer temperatures, and direct mortality in unscreened irrigation diversions. Some limited sources of reduced survival may be changed over a relatively short time (<5 years) (e.g., screening diversions, eliminating water withdrawls), whereas other factors (e.g., excessive fine sediments, stream width to depth ratio) will require a longer time period to respond to restoration activities (10-20 years). For other streams where there is little potential, the sources of habitat degradation have already been eliminated and the system is in a state of long-term recovery, or the stream is located in high quality wilderness or other protected areas. Further, given the large degree of improvement in survival that is needed overall (across the life cycle) and for all listed stocks, it appears unlikely that the range of potential habitat improvement available could elicit an increase in overall survival sufficient to recover these fish.

Delayed hydrosystem mortality

Substantial evidence supports the existence of delayed mortality for Snake River spring and summer chinook and links delayed mortality to hydrosystem experience. This evidence comes in the form of published literature, indirect evidence from life-cycle modeling, and direct evidence from fish-tagging experiments. Further, the compilation of evidence that links hydrosystem experience to delayed mortality has implications for the analyses of management options considered for the recovery of Snake River salmon and steelhead, especially the outcome of a dam breach option. In those analyses, dam breach is likely to dramatically increase the survival of Snake River chinook and lead to eventual recovery of these stocks if delayed mortality is related to hydrosystem experience. Dam breach is predicted to have less of an effect and may be insufficient for recovery if delayed mortality is unrelated to hydrosystem experience. Regardless of the action that is chosen to recover these stocks, the amount of hydrosystem mortality that needs to be compensated must include both direct and delayed components. Based on the evidence summarized in Budy et al. (*in review*), it seems implausible to conclude that little or none of the delayed mortality of Snake River fish is related to the hydrosystem.

Although the results vary some among approaches, all available science appears to suggest that dam breach has the greatest biological potential for recovering Snake River salmon and steelhead.

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