PATH Final Report for Fiscal Year 1998

Plan for Analyzing and Testing Hypotheses (PATH)

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1.0 Executive Summary

1.1 Objectives of PATH

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

The objectives of PATH are to:

1. determine the overall level of support for key alternative hypotheses from existing information, and propose other hypotheses and/or model improvements that are more consistent with these data (retrospective analyses);
2. assess the ability to distinguish among competing hypotheses from future information, and advise institutions on research, monitoring and adaptive management experiments that would maximize learning; and
3. advise regulatory agencies on management actions to restore endangered salmon stocks to self-sustaining levels of abundance (prospective and decision analyses).

PATH products are reviewed by an independent Scientific Review Panel (SRP).

1.2 PATH Accomplishments During Fiscal Year 1998

PATH has made significant progress on all three of these objectives during FY1998. Highlights of PATH activities during the last year include:

- a workshop in October 1997 to evaluate and refine preliminary prospective analyses for spring and summer chinook
- publication of the *Preliminary Decision Analysis Report on Snake River Spring/Summer Chinook* [Marmorek and Peters (eds.)] in March 1998
- publication of *Retrospective and Prospective Analyses of Spring/Summer Chinook Reviewed in FY1997*, in April 1998
- development and refinement of fall chinook passage and life-cycle models, and assembly of fall chinook spawner-recruit data, during February-July 1998
- a preliminary assessment of the effects of management action on fall chinook in August, 1998
- revised *Executive Summary of the Preliminary Decision Analysis Report*, distributed to the Implementation Team on August 4, 1998
- the PATH Weight of Evidence Process to compile and assess the evidence for and against key hypotheses affecting the spring/summer decision analysis during May to August, 1998
- publication and SRP review of the *PATH Weight of Evidence* report (WOE) in August 1998

- a workshop with the SRP to document their best judgements on the relative likelihood of key hypotheses, and a workshop report published in September 1998. This report also included SRP recommendations to PATH regarding the application of experimental management and relevant modeling approaches (objective 2)
- assessment of additional actions both spring/summer chinook and preliminary assessment of options for fall chinook during September-October 1998
- completion of qualitative assessments of the effects of actions on Snake River steelhead (March - October 1998)
- development of historic assessments of SARs (smolt to adult returns) for Snake and Upper Columbia steelhead, and Snake River spring chinook (December 1997 to May 1998)
- initiated assessments on sockeye salmon (October 1998)
- completion of a discussion paper on applying experimental management to the Columbia River, which builds on the SRP's suggestions in their report from the Weighting Workshop (October 1998)

1.3 Summary of Results of Assessments of Actions

1.3.1 General Approach

PATH retrospective analyses have helped to bring a substantial set of empirical information to bear on alternative hypotheses to explain recent declines in Snake River chinook, and have led to considerable improvements in both our understanding and modeling approaches. In addition, there has been considerable convergence on the historical data sets to use in calibrating and testing models, and on many of the assumptions to be made when projecting future population changes.

The PATH retrospective analyses have also highlighted some major uncertainties in past and current conditions that have yet to be resolved because of incomplete data and differences in interpretation. These uncertainties, along with uncertainties in projecting future conditions, imply that a single management action can have a number of possible outcomes, depending on what is assumed about past, present, and future conditions. This range of possible future outcomes of management actions is best captured by modeling salmon populations under a set of alternative hypotheses about uncertain components of the system.

PATH uses decision analysis techniques as a structured framework for looking systematically at the outcomes of management actions under several alternative hypotheses about biological mechanisms that link actions to possible outcomes. Management actions can then be evaluated on the basis of their outcomes. This approach was recommended by the SRP and by independent scientists within PATH as a tool for explicitly considering uncertainties in the decision-making process, in recognition that decisions cannot wait for all uncertainties to be resolved. Decision analysis is not intended to provide a single answer about stock responses to specific actions; rather, it will show which actions are most robust (or risk averse) to the uncertainties captured in quantitative models. The SRP has also recommended an experimental management approach to further reduce remaining uncertainties.

PATH has developed a quantitative decision analysis framework for spring/summer chinook and a preliminary framework for fall chinook. We have developed a qualitative analysis for steelhead using comparisons of the likely effects of actions on spring/summer chinook as a guide to the probable response of steelhead. We have recently begun to consider how our findings might apply to sockeye.

1.3.2 Management Actions

The PATH decision analysis, under the direction of the Implementation Team, has been focused on the extent to which alternative hydrosystem actions can contribute to preventing extinction and aiding recovery of stocks either listed or proposed for listing, including wild spring/summer chinook, fall chinook and steelhead stocks in the Snake River and mid-Columbia region. PATH is focussed on providing a detailed assessment of hydrosystem alternatives, as called for in the NMFS 1995 Biological Opinion. The effects of habitat and harvest management actions are considered in sensitivity analyses (Section 2.3), with further sensitivity analyses being considered in FY99. We consider the possible effects of current hatchery operations, but do not consider major changes in production levels. We also intend to explore options for an experimental management approach, which varies management actions over time and space (including habitat, harvest and hatchery actions) to test key hypotheses and reduce remaining uncertainties. A discussion of experimental management is included in Chapter 6 of this report.

Table 1.3.2-1 shows the range of alternative hydrosystem actions that have been put forward for consideration. In accordance with the priorities on these actions established by the I.T., we have evaluated seven of these:

A1 – current hydrosystem operations (under the 1995 Biological Opinion Interim Action)

A2 – A1+ maximize transportation (without surface collectors)

A2’ – A1+ maximize transportation using surface bypass collectors

A3 – natural river drawdown of the four lower Snake River dams (Lower Granite, Little Goose, Lower Monumental and Ice Harbor)

A6 – in-river option (no transportation, no drawdown, flow augmentation as in A1 plus 1 million acre-feet from upper Snake River, and surface bypass). This option has not yet been fully developed, so we have done a preliminary qualitative assessment of its probable effects on spring/summer chinook, relative to the other actions. Further analysis for fall chinook and spring/summer chinook is under consideration.

A6` – A6, but with flow augmentation as in A1, reduced by 427,000 acre-feet

B1 – natural river drawdown of the four lower Snake River dams **and** John Day dam

Table 1.3.2-1: Hydro system management actions examined by PATH. The A6 and A6‘ options have not yet been quantitatively defined to the same extent as the other options.

Scenario	Flow Augmentation		Drawdown of four Snake River Dams	Drawdown of John Day Dam	Transportation	Major system improvements (1)
	Columbia	Snake				
A1	X	X	-	-	X	- (2)
A2	X	X	-	-	X	- (3)
A2’	X	X	-	-	X	X
A3	X	X	Natural River	-	-	-
A6	X	X(4)	-	-	-	X
A6’	X	X(5)	-	-	-	X
B1	X	X	Natural River	Natural River	-	-

(1) Major system improvements include extended screens and/or surface bypass and/or gas abatement and/or increased spill.
(2) A1 uses current transportation rules.
(3) A2 maximizes transportation using current system configuration.
(4) Dworshak water plus 1.427 million acre-feet from Snake River.
(5) Dworshak water, but no additional Snake River water.

1.3.3 Uncertainties in the Response of Populations to Management Actions

The response of fish populations to hydrosystem management actions under consideration is determined by the hypothesized effects of these actions, and of external environmental influences, on fish at all stages of their life cycle. For spring/summer and fall chinook, we have identified specific alternative hypotheses about:

- factors that affect survival of juveniles through the hydrosystem;
- timing and magnitude of the effects of drawdown on juvenile survival; and
- factors that affect survival of fish outside of the hydrosystem, including climate, harvest and habitat conditions.

Many of these uncertainties cannot be resolved with existing information. The SRP recommended that PATH assess the benefits and risks of an experimental management approach in reducing the key remaining uncertainties. Uncertainties are considered in a less explicit way for steelhead and sockeye.

1.3.4 Performance Measures Used to Assess the Outcomes of the Options

The outcomes of alternative hydro management actions are presented in terms of various measures of how well they perform, both relative to each other and with respect to absolute criteria. Because the primary goal is to determine the hydrosystem actions that should be taken to prevent extinction and lead to recovery of endangered stocks, we focus here on the National Marine Fisheries Service (NMFS) jeopardy standards that were considered in the 1995 Biological Opinion. These standards provide an indication of the ability of actions to increase the spawning abundance of stocks to levels that will avoid extinction and lead to recovery, over short (24 years) and longer (48- and 100-year) time periods. The standards are described in detail in Appendix D of the *PATH Preliminary Decision Analysis Report*, and summarized below.

NMFS Jeopardy Standards

The way in which a specific hydrosystem action affects the chance of an individual spawning stock going extinct is difficult to estimate, because there may be unpredictable population behaviors at low abundance. The performance measure we use to describe the possibility of extinction here is called a "Survival" standard. This was developed by the Biological Requirements Working Group (BRWG 1994), and has largely been accepted by NMFS for use in Snake River chinook salmon jeopardy determinations. The Survival standard is the fraction of time during many simulations that the spawning abundance of a stock is above a specified low threshold. For the seven spring/summer chinook stocks we examined in the Snake River Basin, the threshold level used is either 150 spawners or 300 spawners depending on the characteristics of the stock and the stream. These levels were chosen because below these levels, spawner/recruit relationships are poorly known and unpredictable changes in population behavior are likely to occur. For Snake River fall chinook (one stock only) a provisional survival standard of 300 spawners was developed by the BRWG, and adopted by NMFS in their 1995 Biological Opinion. The survival standard is calculated for simulations run over 24 and 100 years. Survival thresholds were developed by the BRWG specifically for spring/summer chinook, and provisionally for fall chinook, but have not yet been extended to steelhead or sockeye. We therefore use simpler approaches for steelhead and sockeye.

The effect of a certain hydrosystem action on the chance of a spawning stock recovering is described by the "Recovery" standard chosen by the BRWG, who proposed 24- and 48-year recovery standards. The 1995 Biological Opinion used only the 48-year recovery standard: this is the fraction of simulation runs for which the average spawner abundance over the last 8 years of a 48-year simulation is greater than a specified level. For spring/summer chinook stocks the specified level of abundance (the recovery level) is different for each stream, and is 60% of the pre-1971 brood-year average spawner counts in each stream. We use the average abundance of spawners over the last eight years as an index of escapement to compare with the specified recovery level for each stock.[1] For fall chinook, the recovery standard used in the NMFS 1995 Biological Opinion was 2500 spawners. To date no recovery standards have been defined for steelhead or sockeye.

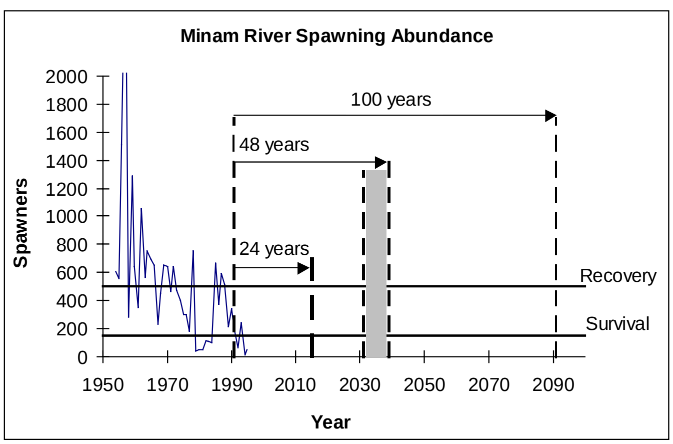


Figure 1.3.4-1: Recent trends in Minam River spawning abundance to 1991, relative to its survival (150) and recovery (450) levels of spawners under NMFS jeopardy standards. Also shown are the 24-, 48-, and 100-year periods for future projections.

[1] We compare the *geometric mean* of simulated future escapements with the arithmetic mean of historical abundances (recovery standard). This difference in summary statistics is recognized, but we use this method because the recovery levels are generally accepted targets, and the geometric mean is an accepted summary statistic for skewed distributions such as abundances of fish over time.

Both jeopardy standards apply to individual stocks. However, the overall performance of the system under different options needs to be described in terms of how each option affects a representative sample of all listed stocks in an Evolutionarily Significant Unit (ESU). To apply these performance standards to a number of stocks, NMFS has defined an overall Jeopardy Standard which considers, among other things, these model-derived probabilities as measures of the ability of an action to prevent extinction of an endangered stock. To meet this standard, an action must result in a "high percentage" of available populations having a "high likelihood" of being above the survival threshold level and a "moderate likelihood" of being above the recovery level. "High" and "moderate" likelihoods have been informally defined as being 0.7 for survival standards, and 0.5 for recovery standards. NMFS has defined "high percentage" of stocks as 80% of the available populations. For the cases in which we are focused on the seven Snake River index stocks, this means that for an action to be considered to have met the overall jeopardy standard, the action must result in six stocks having a probability of 0.7 or greater of being above the survival threshold and a probability of 0.5 or greater of being above the recovery threshold.

Actions can be ranked according to their relative performance (i.e., actions with high probabilities of meeting the standards have greater biological benefits than actions with low probabilities), or according to some criterion (e.g., actions must have at least a 0.50 probability of meeting all of the standards). The establishment of such a criterion is a question for policy-makers, and we have not attempted to define one here.

Box 1-1 outlines how we compute and display the probability that a given action will meet one of the three NMFS standards. Since there are three standards, there are three such probabilities. The overall probability of meeting all three NMFS standards is determined by the lowest of the three probabilities. The following section summarizes these overall probabilities for each action and species. Actual calculation of the probabilities has only been completed for spring/summer and fall chinook, for all actions except for A6 and A6'. Results presented for A6/A6' and for steelhead and sockeye are based on qualitative comparisons, as described in the main report.

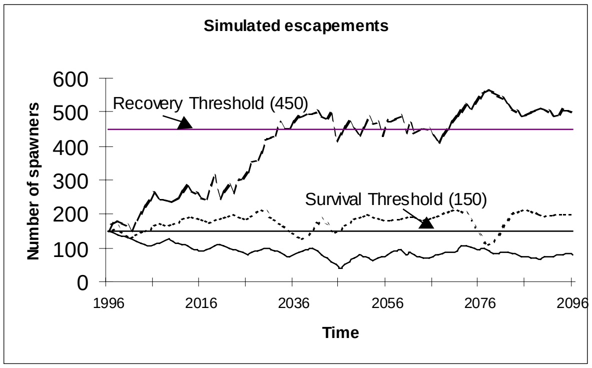
An important point to note about the probabilities of meeting the standards is that these probabilities explicitly incorporate the uncertainties we have defined (Box 1-1). That is, the probabilities are based on outcomes arising from all of the alternative hypotheses and the various combinations of those hypotheses. Therefore, actions with high probabilities meet the standards under a broad range of possible hypotheses about future conditions (a robust action), while actions with low probabilities meet the standards under a narrower range of hypotheses.

The probabilities also incorporate weights on the alternative hypotheses that reflect the relative likelihood of being true. Outcomes derived from hypotheses that have a high likelihood of being true contribute more to the overall probability of meeting the standards than outcomes that are derived from hypotheses with a lower likelihood of being true. Weights on hypotheses can be developed through a comprehensive review of the evidence for and against alternative hypotheses, as we have done for spring/summer chinook through the Weight of Evidence process. However, in the absence of such process, the best we can do is to place equal weights on all of the hypotheses to reflect our lack of knowledge about which hypotheses are more likely than others. This is the case for fall chinook, because we have not yet gone through a Weight of Evidence type of process for that species.

Box 1-1: General steps involved in calculating the probability that a given hydrosystem action will meet a NMFS standard. This example assumes all hypotheses are equally weighted.

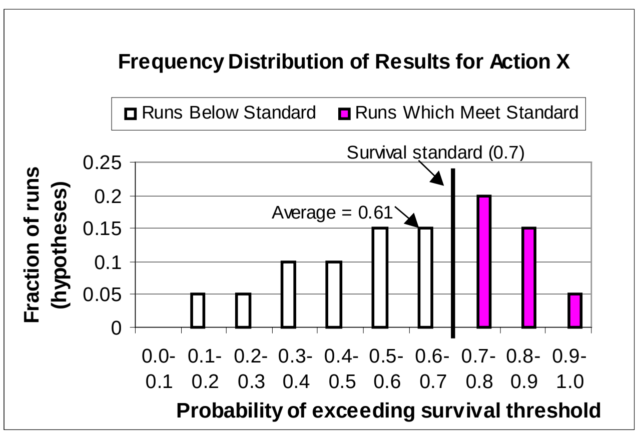
1. Select one combination of hypotheses for this action (a "run").

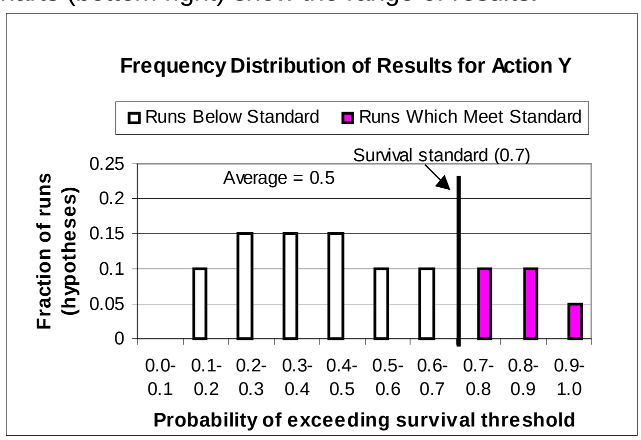
2. Simulate many possible future trajectories for this combination of hypotheses, over the next 100 years, given uncertainties in stock productivity, climate etc.

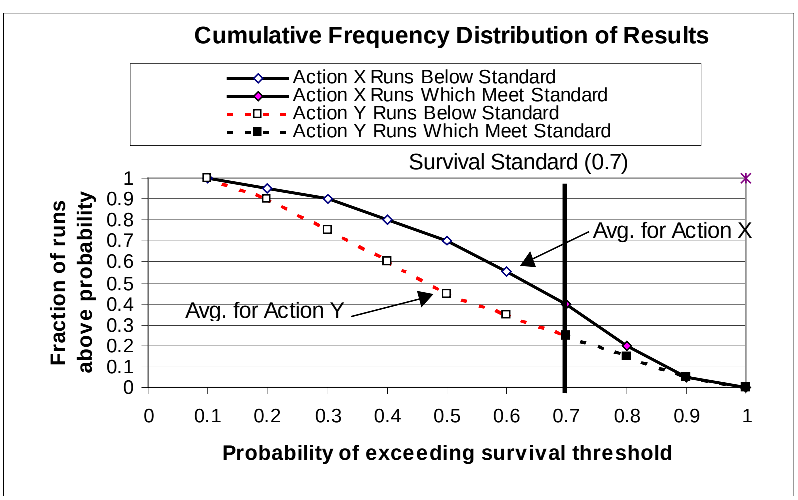


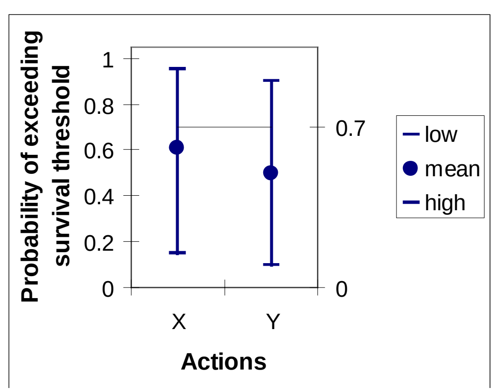
3. Calculate the probabilities of exceeding survival threshold (over 24 and 100 years) and recovery threshold (years 17 to 24, and years 41 to 48).

4. Repeat steps 1-3 for all possible combinations of hypotheses and actions. We want to know the fraction of runs meeting the NMFS standard, and the average probability of exceeding the threshold. These can be displayed in a number of ways. For action X, the fraction of runs meeting survival standard = 0.4 (0.2+0.15+0.05), and average probability of exceeding survival threshold is 0.61. For action Y, the fraction of runs meeting survival standard = 0.25 (0.1+0.1+0.05), and average probability of exceeding survival threshold is 0.5. Cumulative frequency distributions (bottom left) show the fraction of runs above any standard for actions X and Y, and box and whisker charts (bottom right) show the range of results.









1.3.5 Summary of Overall Results for All Species

The hydrosystem management actions were evaluated across a broad range of uncertainties. The natural river actions (A3, B1) exhibited the most robust response across these uncertainties (i.e., those considered to date). For all species, A3 and B1 produce higher biological benefits than the other actions (the rank order of A3 and B1 depends on the delay in implementing Snake River drawdown).

Overall results for all species are summarized in Table 1.3.5-1 and Figures 1.3.6-1 and 1.3.6-2. This summary of results shows the overall assessments of actions, but the results are based on a much more detailed set of assumptions and calculations. We urge the reader to read the remainder of the report to fully understand how the summary results were derived. In general, the relative performance of different actions is a more reliable and consistent outcome of our analyses than the absolute probabilities, which are more sensitive to different assumptions.

We stress that the analyses for spring/summer chinook have been much more extensive and detailed than our fall chinook analyses. For spring/summer chinook we had spawner/recruit data for multiple upstream stocks, directly applicable transport studies, and many years of juvenile and smolt-to-adult survival rate estimates. The SRP reviewed numerous retrospective and prospective products over a period of two years, prior to PATH performing a detailed decision analysis, using two fundamentally different life-cycle models. We produced *a Weight of Evidence* report with 25 separate submissions to evaluate alternative hypotheses, and the SRP performed a formal weighting process. In contrast, for fall chinook, we have only a short time series of juvenile passage data available, no directly applicable transport study results, and a shortage of data on downstream stocks for use as controls in life-cycle modeling. PATH has only worked on fall chinook for only about six months, and the SRP has not yet reviewed the fall chinook work products.

For spring/summer and fall chinook, the values reported are the overall probabilities of meeting all of the standards (fractions of runs), as described in Box 1-1. We report the probabilities for spring/summer chinook with all hypotheses equally weighted (first row), and using the mean of the weights developed by the SRP through the weight of evidence process (second row). Probabilities for fall chinook are with all hypotheses equally weighted. A1 was not run for fall chinook because the A1 system configuration and level of transportation was virtually identical to that of A2. Results for A3 for spring/summer chinook are reported separately for either a 3-year or an 8-year delay before the Snake River dams are removed. Both options were explored for fall chinook as well, but only the mean of these two values is reported here. Results for B1 on average assume a 5.5-year delay before removal of Snake River dams (average of 3 and 8 years), and a 12.5-year delay before removal of John Day dam (average of 10 and 15 years).

One of the reasons for considering multiple species in evaluating the effects of management actions is to uncover any situations where an action may be preferred for one species but is detrimental to another. In reviewing the results summarized in the table above, there do not appear to be any of these situations. Again, we leave it to policy-makers to decide whether the biological benefits of any of these actions are "high enough", given other factors that may influence the final decision.

Table 1.3.5-1: Summary of results for all species. Numbers for spring/summer and fall chinook are overall probabilities of meeting all three NMFS standards, computed from the fraction of runs or hypotheses which met all three standards. Actions with high probabilities meet the standards under a broad range of hypotheses about future conditions (a robust action), while actions with low probabilities meet the standards under a narrower range of hypotheses. These results consider only those management actions and uncertainties in the other H's (habitat, harvest, hatcheries) described in Sections 1.3.2 and 1.3.3.

<table>
<tr><th rowspan="2">Species</th><th colspan="6">Actions</th></tr>
<tr><th>A1</th><th>A2</th><th>A2'</th><th>A6/A6'</th><th>A3</th><th>B1</th></tr>
<tr><td>Spring/summer chinook (equal weights)</td><td>0.35</td><td>0.35</td><td>0.37</td><td rowspan="2">A6: 60 to 150% of A2 performance (wide range of assumptions)
80 to 100% of A2 performance ("more realistic" assumptions).

A6': performs worse than A2.</td><td>0.63 (3-year)
0.47 (8-year)</td><td>0.59</td></tr>
<tr><td>Spring/Summer chinook (SRP weights)</td><td>0.27</td><td>0.25</td><td>0.29</td><td>0.65 (3-year)
0.48 (8-year)</td><td>0.62</td></tr>
<tr><td>Fall chinook</td><td>n/a</td><td>0.15</td><td>0.23</td><td>Analysis not yet completed</td><td>1.00</td><td>1.00</td></tr>
<tr><td>Steelhead</td><td colspan="6">Relative performance of actions for spring/summer chinook applies to steelhead(1)</td></tr>
<tr><td>Sockeye</td><td colspan="3">Less likely to lead to recovery than for spring/summer chinook</td><td colspan="3">Analysis not yet completed</td></tr>
</table>

(1) Actions that meet standards for spring/summer chinook are likely to meet standards for steelhead; actions that do not meet standards for spring/summer chinook may or may not meet standards for steelhead.

1.3.6 More Detailed Performance Measures

Figures 1.3.6-1 and 1.3.6-2 summarize the weighted average probabilities of exceeding specific NMFS survival and recovery thresholds, for both spring/summer chinook and fall chinook. The probabilities of exceeding recovery thresholds show greater discrimination among actions than the survival jeopardy probabilities, for both groups of spring/summer and fall chinook. For spring/summer chinook, the mean SRP weights generated results very similar to applying equal weights (Figure 1.3.6-1). The first part of these charts (i.e., Figures 1.3.6-1a and 1.3.6-2a) show weighted **average** probabilities; the range of probabilities of exceeding survival/recovery thresholds is shown in Figures 1.3.6-1b and 1.3.6-2b.

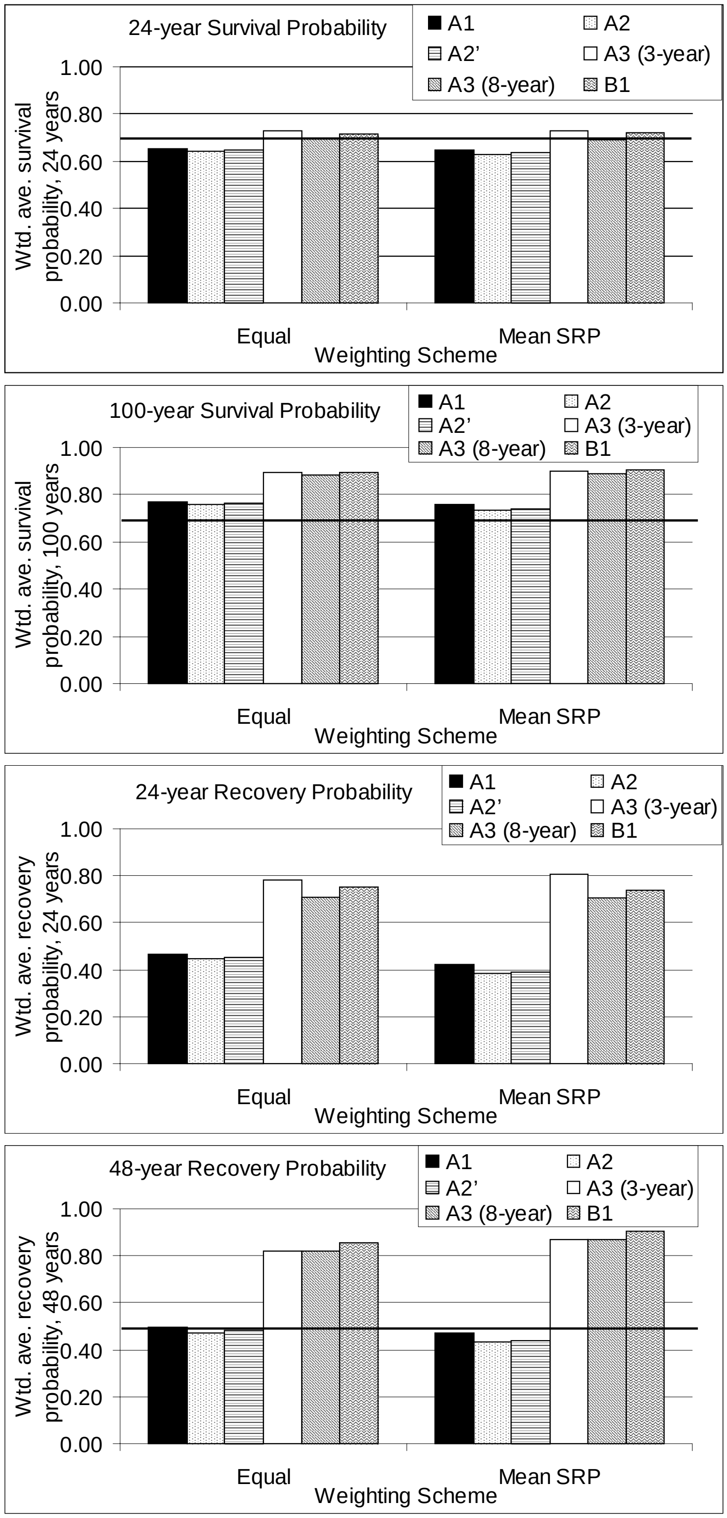


Figure 1.3.6-1a: Average probabilities of exceeding survival and recovery thresholds for spring/summer chinook, using both equally weighted and SRP-weighted hypotheses. Horizontal lines indicate NMFS standard (none for the 24-year recovery probability). These results consider only those management actions and uncertainties in the other H's (habitat, harvest, hatcheries) described in Sections 1.3.2 and 1.3.3.

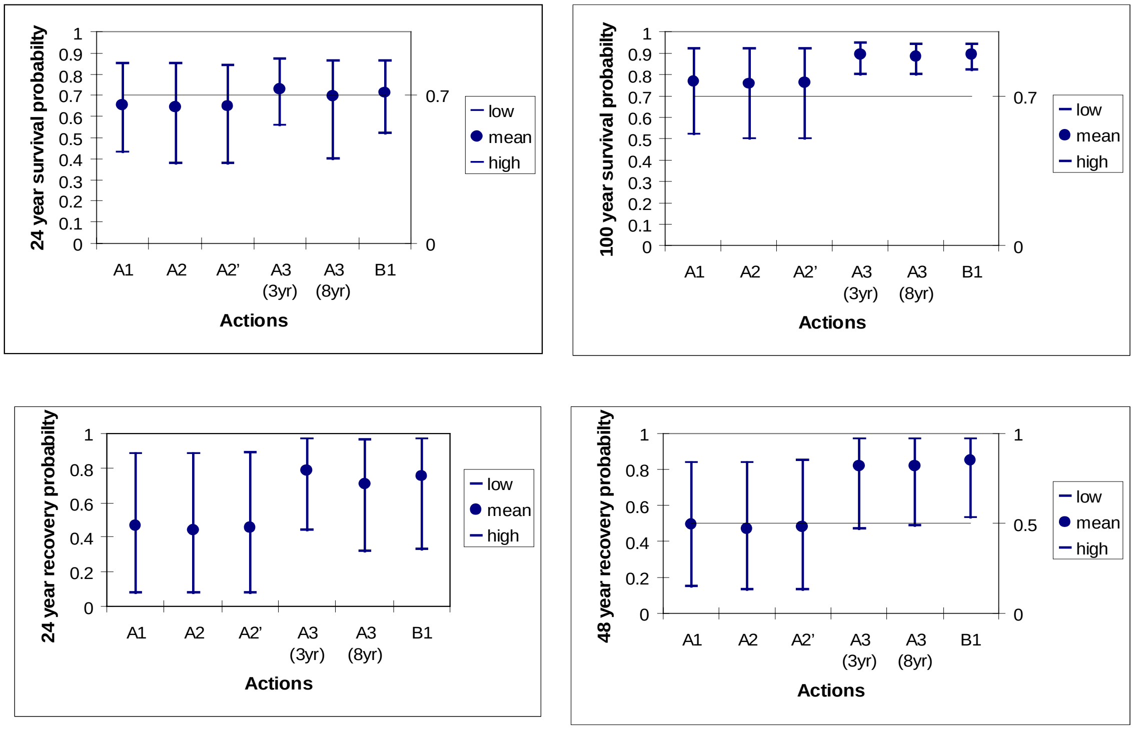


Figure 1.3.6-1b: Lowest, mean and highest probabilities of exceeding NMFS thresholds for survival (top two graphs) and recovery (bottom two graphs) for spring/summer chinook, under six different hydrosystem management actions. Horizontal lines are NMFS standards (none for 24-year recovery probability). Means are calculated weighting all hypotheses equally, and are comparable to those on the left side of Figure 1.3.6-1a. Ranges are unaffected by the assigned weights. These results consider only those management actions and uncertainties in the other H's (habitat, harvest, hatcheries) described in Sections 1.3.2 and 1.3.3.

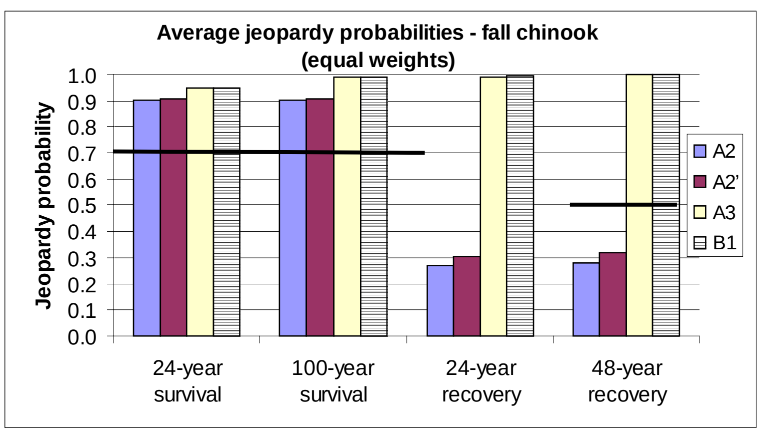


Figure 1.3.6-2a: Average jeopardy probabilities for A2, A2', A3 and B1, using equal weights on all hypotheses.

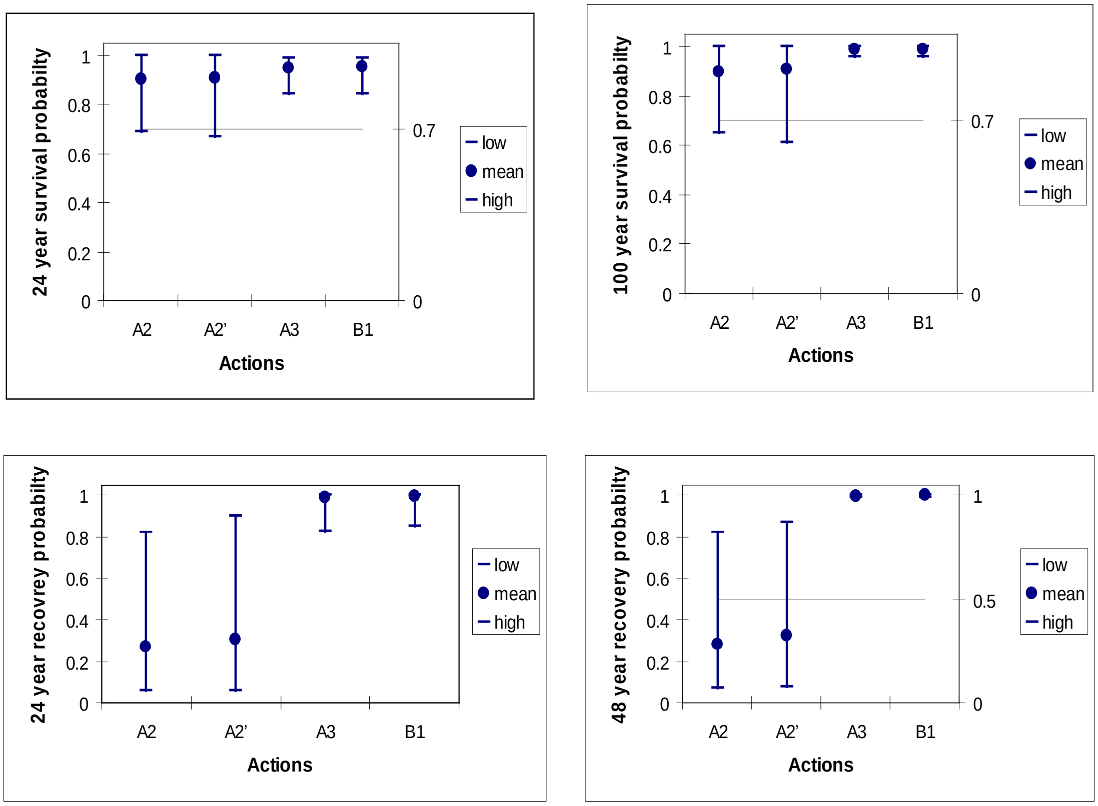


Figure 1.3.6-2b: Lowest, mean (average) and highest probabilities of exceeding NMFS thresholds for survival (top two graphs) and recovery (bottom two graphs) for fall chinook, under four different hydrosystem management actions. Horizontal lines are NMFS standards. Means are computed weighting all hypotheses equally, and are the same as values shown in Figure 1.3.6-2a. These results consider only those management actions and uncertainties in the other H's (habitat, harvest, hatcheries) described in Sections 1.3.2 and 1.3.3.

Smolt to adult survival rates (SARs) estimate survival rates of fish from the time they pass the upper-most dam as smolts to the time they return to that dam as adults. The PATH life-cycle model calculates a median SAR based on many thousands of simulations over a 100-year simulation period. All actions generate SARs that exceed historical estimates (Table 1.3.6-1). Median SAR model estimates to the upper dam were higher for action A3 than for actions A1 and A2. For A3, the median SAR was 4.0%, compared to 2.4% for A1 and 2.3% for A2. Minimum SARs for the three actions were 1.6%, 1.6% and 2.4% for A1, A2 and A3, respectively. Maximum SARs for the three actions were 4.9%, 4.8% and 7.1% for A1, A2 and A3, respectively.

Table 1.3.6-1: Range and median SARs for spring/summer chinook under actions A1, A2, and A3, compared to historical estimates for 1977 to 1994.

	Minimum SAR	Median SAR	Maximum SAR
Historical (1977-1994)	0.2	1.0	2.6
A1	1.6	2.4	4.9
A2	1.6	2.3	4.8
A3	2.4	4.0	7.1

The range of forecasted escapements rapidly increases over the first 30 to 40 years, and then levels out. Figure 1.3.6-3 shows an example of this pattern for one spring/summer chinook stock, Johnson Creek, under different actions. Johnson Creek is frequently the sixth best stock and therefore its escapement values are relevant to the NMFS jeopardy standards. Note that in Figure 1.3.6-3 we have included historical escapement estimates; these past estimates are just one of many possible time sequences which could have occurred. The summary statistics on future projections are summarized from many thousands of possible future trajectories.

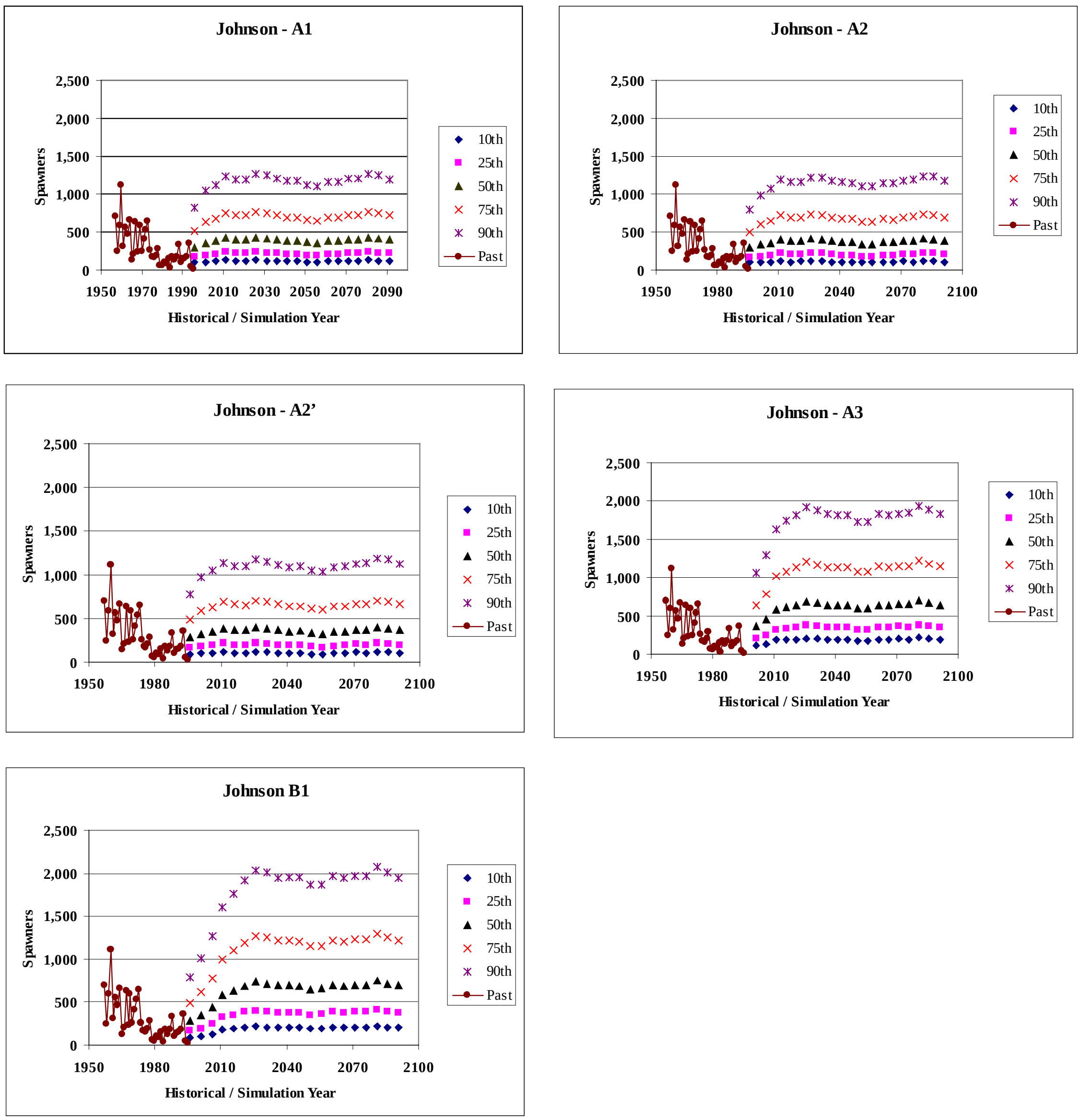


Figure 1.3.6-3: Forecasted range of escapements for Johnson Creek, under different hydrosystem management actions. The points represent the 10th, 25th, 50th, 75th, and 90th percentiles of the forecasted range of escapements in each future year. The past estimates are just one of many possible stock trajectories which could have occurred. The summary statistics on future projections are summarized from many thousands of possible future trajectories. This graph of spawning escapements is not directly comparable to NMFS standards for survival and recovery.

1.3.7 Summary of Sensitivity Analyses

In addition to the uncertainties that are explicitly incorporated into the calculation of probabilities of meeting the standards, PATH has explored the effects of other assumptions on overall results. Many of these have been documented in previous PATH reports. In this report, we have looked specifically at the sensitivity of results to the four factors: habitat, harvest, bird predation in Columbia River estuary, and upstream survival rates. General results are summarized below; more detailed explanation of these factors and their effects on results are in Sections 2.3 and 3.4 of the main report.

1. Management of freshwater spawning and rearing habitat (spring/summer chinook only)

General result: Alternative habitat scenarios (all practical measures taken to protect and restore fish habitat) lead to increases in average jeopardy probabilities for some stocks (Bear Valley, Johnson Creek, Poverty Flat), reductions for others (Marsh Creek, Minam, Sulphur Creek), and mixed results for Imnaha. It is at first surprising that an improvement in habitat for one stock could reduce the abundance of several stocks. There is however a logical explanation. When habitat improvements lead to larger escapements for stronger stocks, this triggers higher in-river harvest rates for all Snake River stocks, including the weaker stocks, since in-river harvest rates increase as total Snake River abundance increases. As a consequence, all stocks are harvested at a higher harvest rate which can lead to lower escapements than would otherwise be the case. This is particularly true for weaker stocks in good habitat (Marsh Creek, Minam, Sulphur Creek) that have zero probabilities of increasing productivity with the increased habitat protection and conservation scenarios.

Overall effects were minor, and did not affect the ranking of actions. Because the analysis was done with a limited set of runs, definite conclusions about the affect of these habitat scenarios on the overall ability of actions to meet the standards (i.e., the fraction of runs in which all of the jeopardy standards are met) are not possible. However, for situations where average jeopardy probabilities are close to the standards with the base habitat scenario, such effects are possible. An analysis with a complete set of runs is required to fully address this question.

2a. Harvest rate reductions (spring/summer chinook)

General result: In-river harvest rates for Snake River spring/summer chinook proceed in a step-wise fashion depending on their abundance (Table 2.3.2-1). Harvest rates have ranged from 3% to 8% since 1975. The effects of harvest rate reductions depend upon the size of the reduction and the specific step in the harvest rate schedule to which the reduction is applied. Small reductions in harvest rates have minimal effects on the probability of meeting survival and recovery standards. Larger reductions in harvest rate can lead to small improvements in the probability of meeting survival and recovery standards (about 0.01 to 0.03), for actions that produce smaller forecasted numbers of spawners (such as A1 or A2). However, at higher levels of forecast abundance (such as under A3) larger reductions in harvest rates can lead to small decreases in the probability of meeting survival and recovery standards (less than 0.03) due to over-escapement, which results in lower levels of recruitment.

Changes in jeopardy probabilities were not sufficient to change the ranking of actions (A3 still produced higher jeopardy probabilities than A1 or A2). The limited set of runs used in this analysis does not allow general conclusions on whether alternative harvest scenarios affect the ability of actions to meet all of the standards. Although the effects of the reductions in harvest rates on jeopardy probabilities are small, they may be large enough to affect the ability of an action to meet the jeopardy standards. This is more likely to occur in situations where average jeopardy probabilities are already close to meeting the standards with the current harvest schedule (e.g., the weighted average 24-year survival probability is close to the standards of 0.7 for all actions; see Table 2.2.4-2). An analysis of all actions with a complete set of runs is needed to fully assess whether the harvest scenarios affect the ability of actions to meet all of the standards.

2b. Harvest rate reductions (fall chinook)

General Result: Alternative ocean harvest schedules were explicitly considered for fall chinook because total harvest rates (which includes a significant ocean harvest on fall chinook) are higher than those on spring-summer chinook and are therefore potentially a more important factor. (We have not yet explored the effects of alternative in-river harvest schedules.) We looked at three ocean harvest scenarios: Current, Conservative (0.85 times current rates), and Liberalized (current harvest rates times 1.15). The 15% change in age specific ocean exploitation rates was based on the latest draft of the U.S. proposal for US v. Canada to the Pacific Salmon Commission (Draft, February 10, 1998). Note that the proposal is based on impacts to age 4 (adult) fish, but we applied the change to all age classes, which results in a greater change from the existing ocean harvest regime. Results suggest that the ocean harvest uncertainties have minor effects on 48-year recovery standards for A2: there is about a 0.025 increase in probabilities under the Conservative scenario, and a 0.03 decrease in probabilities under the Liberal scenario.

3. Sensitivity to recent sources of mortality (i.e., expanded bird populations) (spring/summer chinook only)

General result: The intent in this analysis is to show the effects of explicitly incorporating sources of mortality that may not be reflected in the historical data. The current set of analyses uses historical spawner and recruit data up to brood year 1990 to estimate overall mortality. There are other sources of mortality, however, that may not be reflected in this data. For example, predation on salmon smolts by Caspian terns and other bird predators in the estuary is hypothesized to have increased dramatically in the late 1980's and early 1990's. We considered two alternative ranges of incremental mortality: 5 to 25%, and 10 to 40%.

The simulated effects of an additional mortality affect all actions relatively equally, and thus do not affect the ranking of actions. The maximum decrease in any jeopardy probability for a 5-25% range of additional mortality was 0.15, and for a 10-40% range was 0.23 (both maximum decreases were for action A1, 48-year recovery probability). The smallest decrease was 0.02 with a 5-25% range of additional mortality and 0.05 with a 10-40% range (both for action B1, 100-year survival probability). Insofar as this limited set of runs is representative of the average of all runs (recall that the runs were selected so as to approximate the weighted average jeopardy probabilities over all runs), additional mortality sources do affect the ability of all actions to meet the 24-year survival standard, and the ability of A1, A2, and A2' to meet the 100-year survival standard. However, an assessment of all of the runs is needed to draw this conclusion with confidence.

4. Adult upstream survival rates following drawdown (spring/summer and fall chinook).

General result: Assuming an increase in adult upstream survival rates following drawdown of John Day dam has minimal effects on overall results, for both sets of chinook. For fall chinook, simulations assume on average a two-fold increase in upstream survival with drawdown. PATH needs to carefully scrutinize these conversion rates to ensure the projected survival improvement under drawdown is reasonable.

1.3.8 Experimental Management

An explicit PATH objective is to *assess how future information can distinguish among competing hypotheses, and to advise agencies on research, monitoring and adaptive-management experiments that can maximize learning*. Adaptive or experimental management has been repeatedly recommended by the SRP in their reviews of PATH products and in their recent report (SRP 1998):

> "The weights assigned by SRP members to the key uncertainties reflect the relative likelihood of the alternative hypotheses, based on the evidence currently available. However, all SRP members commented that in some cases, the empirical evidence on which to evaluate alternative hypotheses was poor or lacking. This is because many events have occurred outside of the temporal and spatial range of historic monitoring programs, and outside of our experience. In the face of this level of uncertainty, the SRP felt that it is unrealistic and imprudent to expect irreversible, long-term decisions to recover stocks because there is little confidence that these actions will have the effects they are projected to have. **However, the SRP strongly cautioned that uncertainty should not be used to justify either delaying action or taking no action at all.** Such a misuse of uncertainty in decision-making is not an acceptable component of responsible fisheries management (United Nations Precautionary Approach). Instead, the SRP noted that the existence of uncertainties points to the need to take actions that:
>
> a) result in the best chance at survival and recovery of stocks; and
> b) generate information to reduce uncertainties and improve future decision-making.
>
> Carefully designed and implemented experimental management actions provide that opportunity."

Although PATH has not yet addressed experimental management in depth, PATH retrospective, prospective and decision analyses have helped define key management uncertainties, and have provided a consistent set of data that can be updated and used to evaluate management experiments. Thus experimental management is now a feasible next step, which would add *learning* to the set of criteria already being used to evaluate proposed management actions. Chapter 6 of this report begins the process of assessing the need for experimental management in PATH. We describe what we mean by experimental management, the advantages it provides managers in reducing key uncertainties, how it differs from scientific research, and the six-step cycle that should be followed. We then describe examples of experimental options developed by the SRP and the PATH Planning Group (e.g., changes in the number of hatchery smolts released, in conjunction with A2 or A3), and the steps necessary for the quantitative evaluation of these options. Finally, we list specific PATH objectives related to experimental management for FY 99. The prioritization of all PATH tasks for FY99 will be determined through discussions between PATH and the Implementation Team.

2.0 Spring/Summer Chinook

2.1 Summary of Work Completed in FY98

PATH seeks to use understanding of the past (retrospective analyses) to forecast the range of possible futures under different management actions (prospective analyses). During FY96 and FY97, PATH completed a very comprehensive set of retrospective analyses on Snake River spring/summer chinook populations. These analyses also used data from lower and upper Columbia stocks for comparative purposes. These results, which have been reviewed by the PATH Scientific Review Panel (SRP), are summarized in three reports: 1) *Final Report on Retrospective Analyses for Fiscal Year 1996* [Marmorek (ed.) and 21 authors. 1996]; 2) *Conclusions of FY 96 Retrospective Analyses* [Marmorek and Peters (eds.) and 24 authors. 1996]; and 3*) Retrospective and Prospective Analyses of Spring/Summer Chinook Reviewed in FY1997* [Marmorek, D.R. and C.N. Peters (eds.) and 23 authors. 1998].

In the latter part of FY97 and during FY98, PATH's work on spring/summer chinook focused on prospective analyses. The major accomplishments are summarized below:

- In October 1997, PATH completed a workshop to evaluate preliminary prospective spring and summer chinook forecasts, refine our decision analysis and prospective modeling approach, and begin to document specific hypotheses in detail. Following this workshop, several revisions were made to the decision analysis and prospective modeling approach.
- In March 1998, PATH released a *Preliminary Decision Analysis Report on Snake River Spring/Summer Chinook* [Marmorek and Peters (eds.)]. This report describes the decision analysis approach used by PATH for spring/summer chinook, the methods of forecasting future stock levels, alternative hypotheses, and preliminary results for three management actions: A1 – current operations; A2 – maximization of transportation without surface collectors; and A3 – drawdown to natural river of the four Snake River projects. The Preliminary Decision Analysis was reviewed by the SRP during March 1998. Further analyses led to some significant corrections to the preliminary results, which were included in a revised *Executive Summary of the Preliminary Decision Analysis Report*, distributed to the Implementation Team on August 4, 1998.
- The *Preliminary Decision Analysis Report* weighted all alternative hypotheses equally; if some hypotheses are more likely than others this can affect the expected outcomes of different management actions. At their meeting on February 19th, 1998, the Regional Forum Implementation Team (I.T.) instructed PATH to develop a process for weighing the evidence applicable to key uncertainties. The PATH Planning Group developed such a process, which was approved by the I.T. at their May 7th meeting.
- From May to August, 1998 PATH developed a *Weight of Evidence Report* (WOE). This report analyzed the results to determine which hypotheses were most important in determining the outcomes of management actions, and presented evidence for and against these key hypotheses. Detailed sensitivity analyses were completed for a number of different factors. The WOE report (completed August 21, 1998) included a 160-page synthesis report, and 360 pages of Submissions from PATH participants.
- In late August and early September, the SRP reviewed the WOE report and Submissions. Based on this information, and their previous two years of work in reviewing PATH products, the SRP developed their best judgements on the relative likelihood of key alternative hypotheses, which were provided independently at a workshop held September 8

to 10th, 1998. These judgements, and other recommendations to PATH regarding modeling approaches and the application of experimental management, are summarized in the SRP's report, *Conclusions and Recommendations from the PATH Weight of Evidence Workshop*.

- During the summer and fall of 1998, PATH participants have jointly developed appropriate assumptions for applying passage and life-cycle models to two additional management actions: A2' (maximizing transportation with surface collectors); and B1 (breaching of the four lower Snake River projects and John Day dam). In addition, PATH has completed a very preliminary analysis of the consequences of actions A6 (in-river passage, no transportation, 1 MAF additional flow augmentation, and improvements to dam passage survival) and A6' (in-river passage, no transportation, no flow augmentation from upper Snake basin). This analysis of A6 did not involve detailed modeling due to lack of time, but was intended to provide a quick picture of the maximum potential benefits of this alternative.

All of the above listed PATH reports are available from the PATH web site: www/bpa.gov/Environment/PATH.

The following section of the report (Section 2.2) provides the forecast outcomes for five actions: A1, A2, A2', A3, and B1, together with the very preliminary results for A6 and A6'. Section 2.3 explores the sensitivity of these outcomes to some alternative assumptions with respect to habitat improvement, harvest rates, and bird predation. The general modeling approach is described below. Though this material also is included in the Weight of Evidence report, it is repeated here for the convenience of readers.

2.1.1 Modeling Approach

The previous PATH retrospective analyses have elucidated a great deal (see PATH 1996 Conclusions Document), and have also pointed out uncertainties in past conditions due to incomplete data and potentially confounding influences (Box 1 in Figure 2.1.1-1). These uncertainties generate a range of alternative assumptions about historical conditions, such as the mortality of fish at specific dams in past years, or the success of past transportation experiments (Box 2). These alternative assumptions about the past, together with historical flow information (Box 3), are used in retrospective modeling analyses that generate quantitative estimates of parameters needed to run models into the future. This requires running both passage models, which estimate survival from Lower Granite Reservoir to Bonneville Dam (Box 4) and life cycle models (Box 6). Spawner-recruit data (Beamesderfer et al. 1997) and environmental data (e.g., climate indicators) are used for calibration of the life cycle models' stock production functions and other parameters (Box 5). The retrospective modeling analysis quantifies our understanding of the variability in survival rates, and the factors which affect them. Results from the retrospective analysis are passed to the prospective analysis (Box 7). The prospective modeling analysis (Boxes 11 and 13) quantifies the range of possible futures, expressed as specific performance measures. This set of possible futures depends on:

- the understanding and estimated parameter values gleaned from the retrospective analysis (Box 7);
- the specific future action under consideration (Box 8; scenarios A1, A2, A2', A3, B1). This set of actions has been developed by the Implementation Team (I.T.), and draws from previous experience of analyzing a much larger set of options (refs: Biological Opinion; System Operating Review; System Configuration Study). The hydrosystem operating requirements associated with each option are described in Appendix C of the Preliminary Decision Analysis Document.

- the expected flows associated with each action (Box 10); and
- assumptions about future conditions, including passage survival assumptions (Box 9) such as fish guidance efficiency through bypasses around dams, and non-passage assumptions (Box 12) such as harvest schedules, habitat improvements and future climate.

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

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

2. A passage model translates the projected set of flows, spills, and dam configurations and operations for a given year into the estimated passage survival of both transported and non-transported smolts through the migration corridor from the head of Lower Granite Reservoir to the tail-race of Bonneville Dam. The passage models simulate passage survival rates under each management action for the water years 1977-1992, to compute the improvement in survival relative to the retrospective period. The longer term water record (i.e., 1929-1988) is considered in step 4. We have used two different passage models, CRiSP and Spring FLUSH, which use different approaches to predicting passage survival rates.
3. One of the key pieces of information passed from the retrospective modeling analysis to the prospective analysis are estimates of the ratio of post-Bonneville survival rates of transported to that of non-transported fish. These ratios are generated by combining estimates of historical passage survival rates with the results of transportation experiments.

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

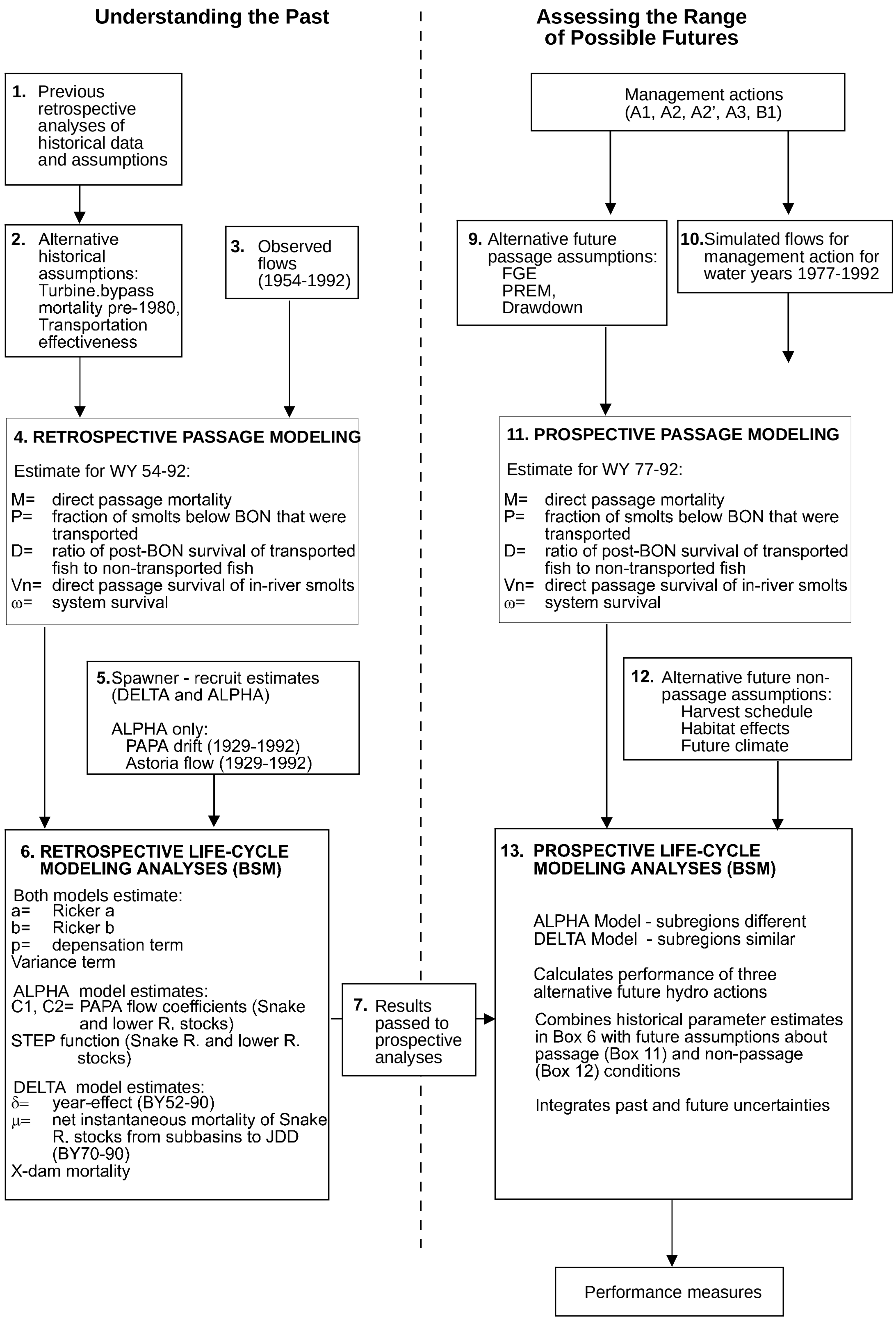


Figure 2.1.1-1: Diagram of analytical approach used in the decision analysis.

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2.2 Prospective Results for Spring/Summer Chinook

This section provides more details on the relative performance of the five Hydro actions under consideration (A1, A2, A2’, A3, and B1) for spring/summer chinook (Table 2.2-1). The organization of this section roughly follows the sequence of analyses PATH undertook to arrive at the overall results presented in the Executive Summary. Section 2.2.1 documents the various assumptions and hypotheses investigated by PATH. Section 2.2.2 shows the range and frequency of results obtained from these combinations of assumptions (we refer to these combinations as “runs”, because each combination represents a distinct and unique run of the life-cycle model). Section 2.2.3 explores the relative effects of the various assumptions and hypotheses on the results. Section 2.2.4 documents the relative weights placed on the most critical assumptions that resulted from the PATH Weight of Evidence Process completed in FY98 and provides further analyses of the weighted results of the prospective analyses. Section 2.2.5 provides a qualitative evaluation of the performance of A6/A6’, an additional pair of actions. This alternative was not developed sufficiently to allow a formal quantitative analysis as was done for A1, A2, A2’, A3, and B1. Finally Section 2.2.6 compares various measures of survival of smolts through the passage corridor. These comparisons can be useful for understanding differences in responses of fish populations to the alternative actions.

Table 2.2-1: Five Hydro actions under consideration for spring/summer chinook.

Scenario	Flow Augmentation		Drawdown of four Snake River Dams	Drawdown of John Day Dam	Transportation	Major system improvements (1)
	Columbia	Snake				
A1	X	X	-	-	X	- (2)
A2	X	X	-	-	X	- (3)
A2’	X	X	-	-	X	X
A3	X	X	Natural River	-	-	-
A6	X	X(4)	-	-	-	X
A6’	X	X(5)	-	-	-	X
B1	X	X	Natural River	Natural River	-	-

(1) Major system improvements include extended screens and/or surface bypass and/or gas abatement and/or increased spill.
(2) A1 uses current transportation rules.
(3) A2 maximizes transportation using current system configuration.
(4) Dworshak water plus 1.427 million acre-feet from Snake River.
(5) Dworshak water, but no additional Snake River water.

Portions of some of the analyses presented in these sections of the FY98 Final Report have already been produced in previous PATH reports, but this is the first time that all five of the actions have been analyzed. Also, the results in this report incorporate the changes in assumptions and models that have occurred since the *Preliminary Decision Analysis Report* was published in March (these changes are described in Appendix B of the *PATH Weight of Evidence Report*).

2.2.1 Assumptions and Uncertainties

The response of fish populations to hydrosystem management actions under consideration is determined by the hypothesized effects of these actions, and of external environmental influences, on fish at all stages of their life cycle. In many cases, there is considerable uncertainty about what these effects and influences will be. This is because of a lack of data (e.g., no direct measurements of survival of fish in the ocean), differing interpretations of existing data, or because we have no experience from which to estimate effects of future actions (e.g., drawdown of Snake River dams) or environmental conditions.

Previous PATH work has helped to resolve some uncertainties, and to clarify others (*FY96 PATH Report on Retrospective Analyses*, *FY96 Conclusions Document*, *FY97 Report on Retrospective Analyses*). Uncertainties from modeling analyses that PATH has not been able to resolve have been incorporated into our analyses of the actions using decision analysis, which provides a structured framework for considering the effects of uncertainties on outcomes of management actions. Much more detail on these uncertainties, as well as on the actions, measures of performance, and analytical techniques used in the decision analysis framework, was published in the *PATH Preliminary Decision Analysis Report* and in the *PATH Weight of Evidence Report*. Here we provide only a brief description of the uncertainties incorporated into the analyses.

There are many uncertainties that can potentially affect the responses of fish populations to management actions. We have focused on fourteen important uncertainties, and have laid out a range of alternative hypotheses for each (alternative hypotheses are defined in Table 2.2.1-1). These uncertainties are:

1. *In-river survival assumptions* – uncertainty in direct survival of in-river fish, and the partitioning of in-river survival between dam and reservoir survival.
2. *Fish guidance efficiency (FGE)* – uncertainty in the effectiveness of extended-length screens in diverting fish away from the turbines, relative to standard-length screens.
3. *Turbine/Bypass Mortality* – uncertainty in historical estimates of bypass and turbine mortality for some projects prior to 1980.
4. *Predator Removal Effectiveness* – uncertainty in the effect of the predator removal program (i.e., removal of squawfish for bounties) on survival of salmon smolts in reservoirs.
5. *Duration of Pre-Removal Period* – the duration of time between a decision to proceed with drawdown and actual removal of dams (pre-removal period) due to uncertainty in the Congressional appropriations process and the possibility of litigation.
6. *Equilibrated juvenile survival rate after drawdown* – uncertainty in the long-term physical and ecological effects of drawdown (e.g., change in density of predators).
7. *Duration of Transition Period* – duration of period between completion of dam removal and establishment of equilibrium conditions in the drawndown section of the river (transition period), reflecting uncertainty in the physical and biological responses to drawdown (e.g., short-term response of predators, release of sediment).
8. *Transportation assumptions* – uncertainty in the relative survival of transported and non-transported fish after the fish have exited the migration corridor (i.e., below Bonneville Dam).
9. *Life-cycle model* – uncertainty in the extent to which Snake River and lower Columbia stocks share common mortality effects.
10. *Extra mortality/Future climate* – Extra mortality is any mortality occurring outside of the juvenile migration corridor that is not accounted for by either: 1) productivity parameters in spawner-recruit relationships; 2) estimates of direct mortality within the migration corridor (from passage models); or 3) for the delta model only, common year effects affecting both Snake River and Lower Columbia River stocks. Extra mortality can in theory occur either before or after the hydropower migration corridor.

    Note on hypothesis about effects of hatchery fish on extra mortality:
    PATH has developed three major alternative hypotheses for extra mortality – BKD (extra mortality will continue into the future irrespective of management action), Hydro[2] (extra mortality is proportional to hydropower-related mortality), and Regime Shift (extra mortality follows a 60-year cycle corresponding to regular climatic cycles). However, in their review of these hypotheses, the Scientific Review Panel noted that these hypotheses could be generalized to include other mechanisms, including hatchery effects. Some SRP members considered hatchery effects as part of hydro effects (because of crowding in barges and forebays, etc.), and assumed that if dams were removed then hatchery production (as a

[2] Results in this report for the Hydro hypothesis are based on the original implementation of the Hydro extra mortality hypothesis, as described in the PATH Preliminary Decision Analysis Report. During the Weight of Evidence process, an alternative formulation of this extra mortality hypothesis was developed. Under this reformulation of the hydro hypothesis, extra mortality is essentially "here to stay" under actions A1 and A2 (i.e., is similar to the BKD hypothesis), and reverts to pre-dam levels when dams are removed under action A3. Although some members of the SRP felt that the new formulation was preferable to the current formulation because it was simpler, we have not yet implemented the new formulation into the full PATH analyses for spring/summer chinook. However, the alternative formulation was used for fall chinook analyses (see section 3.4.1).

mitigative measure) and, consequently, the effects of hatchery fish on wild fish would be reduced. Others assumed that the reversible component of hatchery effects was captured in the hydro hypothesis, while the irreversible component was captured in the BKD hypothesis.

Uncertainty in future climate relates to future patterns in climatic conditions. Extra mortality and future climate are coupled because they are closely linked with one another.

11. *Habitat effects* – uncertainty in the biological effects of future habitat management actions.
12. *Harvest Schedules* – alternative harvest rates for Snake River spring and summer chinook.
13. *Additional sources of mortality* - effects of explicitly incorporating sources of mortality that may not be reflected in the historical data. The current set of analyses uses historical spawner and recruit data up to brood year 1990 to estimate overall mortality. However, there are other sources of mortality (e.g., predation on salmon smolts by Caspian terns and other bird predators) that may not be reflected in this data.
14. *Adult conversion rates following John Day drawdown* – uncertainty in the effects of removing John Day dam on the survival of adult fish migrating upstream through the John Day reach.

Uncertainties 5, 6, and 7 only apply when projecting the effects of drawdown to natural river of the four lower Snake River dams (option A3) and of the four Snake River dams + John Day Dam (option B1). Uncertainty 14 applies only when projecting the effects of drawdown to natural river of John Day dam. Uncertainties 11-14 were considered through sensitivity analyses using a limited set of assumptions (additional sensitivity analyses have been proposed for FY99 – see Section 7).

Table 2.2.1-1: Set of uncertainties and alternative hypotheses considered in this analysis.

Uncertainty	Hypothesis Label	Description
Uncertainties / hypotheses related to downstream passage to Bonneville Dam		
In-river survival assumptions —Passage Models	PMOD1	CRiSP estimates of in-river survival (Vn) and proportion transported
	PMOD2	FLUSH estimates of in-river survival (Vn) and proportion transported
Fish Guidance Efficiency (FGE)	FGE1	FGE w/ESBS > FGE w/STS (values depend on project) (ESBS = extended length submersible bar screens). (STS = standard length submersible travel screens). e.g., LGR 1996-1997: FGE1 = 78%
	FGE2	FGE w/ESBS = FGE w/STS. e.g., LGR 1996-1997: FGE2 = 55%
	FGE3	FGE w/ SBC (A2' only)
Historical / Turbine + Bypass Survival (Note 1)	TURB1	Turbine survival = 0.9. Bypass survival = 0.97 - 0.99, depending on the project. Used for post-1980 years in all runs.
	TURB4	Highest pre-1980 mortality; turbine and bypass mortality are due to descaling alone
	TURB5	Lowest pre-1980 mortality; turbine mortality rate = half descaling rate; bypass mortality rate = descaling rate
Predator removal efficiency	PREM1	0% reduction in reservoir mortality resulting from predator removal program.
	PREM3	25% reduction in reservoir mortality.
Duration of pre-removal period under drawdown	PRER1	3 years – Snake River drawdown 10 years – John Day drawdown
	PRER2	8 years– Snake River drawdown 15 years– John Day drawdown

Uncertainty	Hypothesis Label	Description
Equilibrated Snake River juvenile survival rate under drawdown	EJUV1	Est. survival rate through drawndown lower Snake R. reach = 0.85 Est. survival rate through drawndown John Day reach = 0.90
	EJUV2	Est. survival rate through drawndown lower Snake R. reach = 0.96 Est. survival rate through drawndown John Day reach = 0.98
Duration of Transition Period after drawdown	TJUVa	Survivals reach equilibrated values 2 years after dam removal (Snake River and John Day).
	TJUVb	Survivals reach equilibrated values 10 years after dam removal (Snake River and John Day).
Other uncertainties / alternative hypotheses		
Transportation models (Note 2)	TRANS1 or T1 (FLUSH only)	Relationship between TCR and survival of control fish, based on data from all transport studies conducted at LGR and LGO dams between 1971-1989. This relationship, and FLUSH in-river survival, used to estimate relative post-BONN survival of transported fish (D) in both retrospective and prospective analyses. Prospective and retrospective D values lower than TRANS4 and relatively constant between pre-1980 and post-1980 periods.
	TRANS4 or T4 (CRiSP only)	For pre-1980 retrospective analyses, relative post-BONN survival set at median D-value estimated from seven T:C studies in 1970's and associated CRiSP in-river survival rate estimates. Post-1980 retrospective analyses use median D-value estimated from four T:C studies in 1980's, and CRiSP in-river survivals. For prospective analyses, D-value randomly selected from four post-1980 values. Prospective and retrospective D values higher than TRANS1 and 2; increase in retrospective D values after 1980 reflects improved transport conditions.
Life-cycle model	PROSPA - ALPHA	Extra mortality is specific to each sub-region, and affected by climate variables.
	PROSPD - DELTA	Extra mortality is independent of the common year effects which affect several subregions.
Extra mortality / Future climate	BKD/Markov	Extra mortality is here to stay; prospective D values selected randomly from post-1980 values; future climate is sampled from historical distribution with autoregressive properties.
	BKD/Cyclical	Extra mortality is here to stay; prospective D values selected randomly from post-1980 values; future climate follows cyclical pattern.
	Hydro/Markov	Extra mortality is proportional to hydropower-related mortality, with a different proportionality coefficient in each year. Prospective D values are selected according to water year. Future climate is sampled from historical distribution with autoregressive properties.
	Hydro/Cyclical	Extra mortality is proportional to hydropower-related mortality, with a different proportionality coefficient in each year. Prospective D values are selected according to water year. Future climate follows cyclical pattern, with both long (60-year) and shorter (18-year) cycles.
	Regime Shift/Cyclical	Both extra mortality and future climate follow cyclical pattern. Prospective D values selected randomly from post-1980 values
Habitat Effects (Sensitivity analyses only)	HAB0	No change in stock productivity (Ricker a parameter).
	HABA	Ricker a adjusted to reflect continuation of current habitat management according to existing management plans.
	HABB	Ricker a adjusted to reflect implementation of all possible habitat restoration or protection (original implementation).
	HABC	Ricker a adjusted to reflect implementation of all possible habitat restoration or protection (revised implementation).
Harvest Schedules (Sensitivity analyses only)	Current	Harvest schedule based on existing harvest management plan.
	Conservative	Current harvest rates (for run sizes below MSP level) divided by 1.5
	Liberal	Current harvest rates (for run sizes below MSP level) multiplied by 1.5
	Low	Lowest harvest rate in current schedule applied at all run sizes.

Uncertainty	Hypothesis Label	Description
Additional sources of Mortality (Sensitivity analyses only)	5 to 25%	Additional mortality rate of between 5 and 25% applied in each simulated year
	10 to 40%	Additional mortality rate of between 10 and 40% applied in each simulated year
Adult conversion rate through John Day reach following John Day drawdown (Sensitivity analyses only	No increase	No increase in adult conversion rates through John Day reach following drawdown of John Day dam.
	Increase	Adult conversion rates through john Day reach increase to 1.0 following John Day drawdown.

Notes:

1. All of the results presented in this report are based on the TURB1, 4, and 5 hypotheses. An additional TURB hypothesis (TURB6) was run with the FLUSH model, but sensitivity analyses showed that TURB6 results are intermediate to those of TURB1, 4, and 5 so effects of the TURB6 hypothesis are subsumed by results for the other TURB hypotheses.
2. All of the results presented in this report are based on the TRANS1 (with FLUSH) and the TRANS4 model (with CRiSP). Another transportation model (TRANS2) was run with the FLUSH model. This model tends to result in lower D values. The effects of this transportation model on overall results were explored in Appendix C of the *PATH Weight of Evidence Report*.

2.2.2 Range and Distribution of Results

Measures of the Performance of Actions

Each of the combinations of hypotheses in Table 2.2.1-1 produces a different projected outcome of each management action. PATH has used the NMFS Jeopardy Standards as the primary outcome on which to compare the alternative actions. The standards are a measure of the ability of management actions to promote survival and recovery of endangered chinook stocks. Each action and each combination of hypotheses produces four values:

1. the probability that the number of spawners for a Snake River spring/summer chinook stock will exceed a pre-defined survival threshold level over the first 24 years of the 100-year simulation period (this is referred to as the 24-year survival probability).
2. the probability that the number of spawners for a Snake River spring/summer chinook stock will exceed the survival threshold level over the full 100-year simulation period (this is referred to as the 100-year survival probability)
3. the probability that the number of spawners will exceed a pre-defined recovery threshold level of spawners in the last eight years of a 24-year time period (this is referred to as the 24-year recovery probability).
4. the probability that the number of spawners will exceed a pre-defined recovery threshold level of spawners in the last eight years of a 48-year time period (this is referred to as the 48-year recovery probability)

We use the "Survival" standards to describe the possibility of extinction. These standards were developed by the Biological Requirements Working Group (BRWG 1994), and have largely been accepted by NMFS for use in Snake River chinook salmon jeopardy determinations (the NMFS approach is described in Appendix D of the *Preliminary Decision Analysis Report*). The Survival standard is the fraction of time during many simulations that the spawning abundance of a stock is above a certain specified low threshold. The threshold level used is either 150 spawners or 300 spawners depending on the characteristics of the stock and the stream. These levels were chosen because below these levels, spawner-recruit relationships are poorly known and unpredictable changes in population behavior are likely to occur. This standard is calculated for simulations run over 24 years and simulations over 100 years.

The effect of a certain hydrosystem action on the chance of a spawning stock recovering is described by the "Recovery" standard chosen by the BRWG (see details in Appendix D of the *Preliminary Decision Analysis Report*). The recovery standard is the fraction of simulation runs for which the average spawner abundance over the last 8 years of a 48-year simulation is greater than a specified threshold. The specified level of abundance (the recovery threshold) is different for each stream, and is 60% of the pre-1971 brood-year average spawner counts in each stream. We used the geometric mean abundance of spawners over the last eight years as an index of escapement to reflect the skewed distribution of abundances normally observed over time.[3]

These descriptions are for single stocks, but the overall performance of the system under different options needs to be described in terms of how each option affects a representative sample of all listed stocks in an Evolutionarily Significant Unit (ESU). To apply these performance standards to a number of stocks, NMFS has defined an overall Jeopardy Standard which considers, among other things, these model-derived probabilities as measures of the ability of an action to prevent extinction of an endangered stock. To meet this standard, an action must result in a "high percentage" of available populations having a "high likelihood" of being above the survival threshold level and a "moderate likelihood" of being above the recovery level. "High" and "moderate" likelihoods have been informally defined as being 0.7 for survival standards, and 0.5 for recovery standards. NMFS has defined "high percentage" of stocks as 80% of the available populations. For the cases in which we are focussed on the seven Snake River index stocks, this means that for an action to be considered to have met the overall jeopardy standard, the action must result in six stocks having a probability of 0.7 or greater of being above the survival threshold and a probability of 0.5 or greater of being above the recovery threshold.

If an action produces survival and recovery probabilities that equal or exceed these critical levels, that action is considered to have met all of the jeopardy standards[4]. Whether or not an action meets the jeopardy standards is another measure by which to judge its performance. For each action, however, there are many combinations of assumptions that determine whether these standards are met. Therefore, we express the ability of an action to meet the standards as the fraction of all of the combinations of hypotheses in which these standards are met. A high fraction indicates that the action performs well (i.e., meets the jeopardy standards) under a broad range of uncertainties. A small fraction indicates that the action performs well only if certain hypotheses are assumed to be true, and is therefore less robust to uncertainty.

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[3] We are comparing the geometric mean of simulated future escapements with the arithmetic mean of historical abundances (recovery standard). This difference in summary statistics is recognized, but we use this method because the recovery levels are generally accepted targets, and the geometric mean is an accepted summary statistic for skewed distributions.

[4] The 24-year recovery probability is not an official NMFS Jeopardy Standard but is calculated as an "indicator" of the short-term ability of stocks to achieve recovery levels.

Range and Distribution of Survival and Recovery Probabilities

In total, there are hundreds or thousands of potential combinations of the hypotheses in Table 2.2.1-1 for each action[5]. Each combination (or run) results in a unique outcome for that particular action. The raw results from these hundreds or thousands of outcomes (for the sixth best stock) can be displayed as a cumulative frequency distribution. An example of such a distribution for a single action (A1) and jeopardy standard (the 24-year survival) is shown in Figure 2.2.2-1; cumulative frequency distributions for all actions and standards are shown in Appendix C. We use this method of displaying results because it provides a broad picture of the potential risks and benefits to the stocks of a given action. Two lines are shown this graph; one where all combinations of assumptions are weighted equally ("equal weight"), and another ("mean SRP weight") where each combination is weighted according to the PATH Scientific Review Panel's assessment of the relative likelihood of that combination (see Section 2.2.4 of this report and the SRP's "Conclusions and Recommendations from the PATH Weight of Evidence Workshop" for more details on the weighting process).

The cumulative frequency distribution can be explained as follows. We focus our explanation for now on the "equal weight" line (diamond symbols) on the graph. The range of survival probabilities that resulted from A1 are displayed along the bottom axis. For each of these values, the point on the graph shows the fraction of all of the 240 runs for A1 in which that particular survival probability was equaled or exceeded. For example, 0.84 of all of the runs for A1 produced a 24-year jeopardy probability that was 0.52 or higher (see point on graph labeled "Example"). All of the A1 runs produced 24-year jeopardy probabilities that were greater than 0.43 (i.e., 0.43 is the minimum 24-year survival probability for A1), and none of the runs produced a 24-year jeopardy probability that was greater than 0.85 (0.85 is the maximum 24-year survival probability for A1). The minimum and maximum values for other actions and jeopardy standards are summarized in Table 2.2.2-1.

[5] 240 runs for A1, A2, A2', and B1; 1920 runs for A3

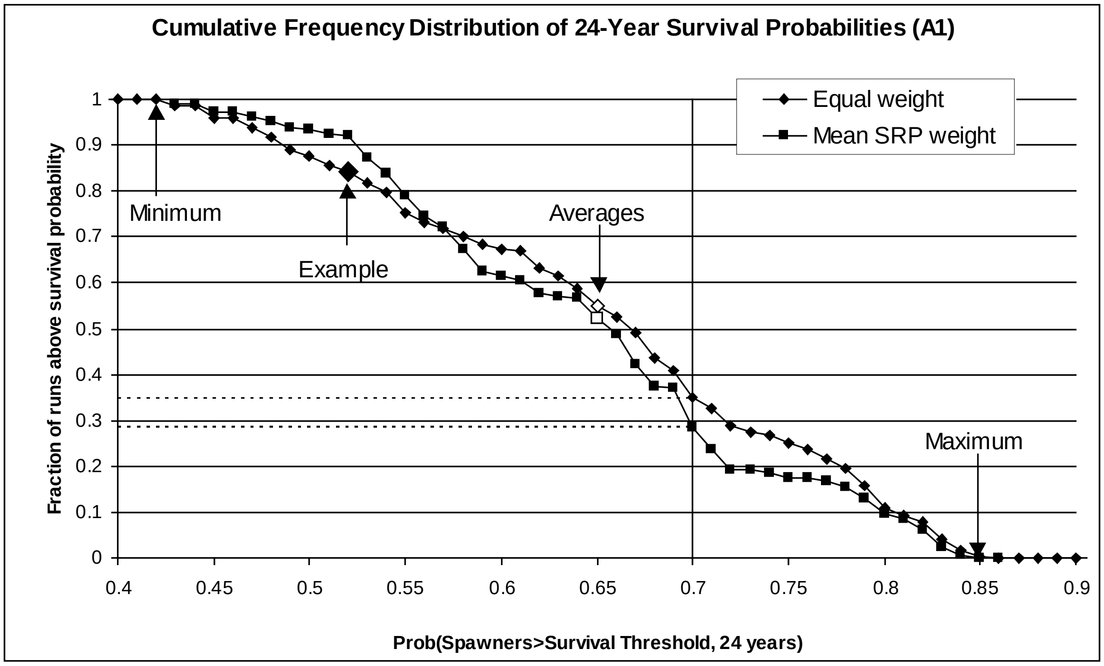


Figure 2.2.2-1: Cumulative frequency distribution of 24-year survival probabilities for Action A1. Points labeled on the graphs show key statistics that are used to summary the results. These points are explained in the text.

Table 2.2.2-1: Minimum and maximum values of survival and recovery probabilities.

	Range of Survival and Recovery Probabilities			
Action	**24-year survival**	**100-year survival**	**24-year recovery**	**48-year recovery**
A1	0.43 to 0.85	0.52 to 0.92	0.08 to 0.88	0.15 to 0.84
A2	0.38 to 0.85	0.50 to 0.92	0.08 to 0.88	0.13 to 0.84
A2’	0.38 to 0.84	0.50 to 0.92	0.08 to 0.89	0.13 to 0.85
A3 (3-year delay)	0.56 to 0.87	0.80 to 0.95	0.44 to 0.97	0.47 to 0.97
A3 (8-year delay)	0.40 to 0.86	0.80 to 0.94	0.32 to 0.96	0.49 to 0.97
B1	0.52 to 0.86	0.82 to 0.94	0.33 to 0.97	0.53 to 0.97

Summary Statistics

Although figures like Figure 2.2.2-1 provide a broad picture of the raw results of the PATH modeling analyses, these results must be summarized further to provide useful information for decision-making. We focus on two summary statistics for comparing the outcomes of actions and the ability of actions to meet the standards. The derivation of these statistics is described here, using Figure 2.2.2-1 for illustration. Section 2.2.4 describes these results in more detail. The first summary statistic is the average of all of the outcomes for a particular action over all of the runs. The average can be calculated where all of the runs are weighted equally, or where the runs are weighted based on their likelihood of being true. The average value of 24-year survival probabilities for A1 using equal weights is shown in Figure 2.2.2-1 as an open diamond; the SRP weighted average 24-year survival probability for A1 is shown in that figure as an open square. For this particular action and jeopardy probability, the equal weight and the mean SRP weighted averages are the same (0.65), but this is not necessarily true for other actions and standards (see Figure 2.2.4-1).

The second summary statistic is the fraction of runs for a particular action that meet the NMFS jeopardy standards. The NMFS jeopardy standards define critical levels for these probabilities (0.7 for the survival standards, 0.5 for the recovery standard) as an absolute level of performance that actions should achieve. As discussed earlier, we express the ability of an action to achieve these standards as the fraction of all of the combinations of hypotheses in which these standards are met. Management actions that have large values meet the standards under a broad range of possible hypotheses about future conditions (a robust action). Management alternatives with low fractions meet the standards under a narrower range of hypotheses.

In Figure 2.2.2-1, the defined critical level for the 24-year standard probability of 0.7 is marked with a vertical line. The point at which the graphs cross this vertical line represents the fraction of all 240 runs for A1 in which this critical level is equaled or exceeded. This fraction with equal weights applied is 0.35; with the mean SRP weight the fraction of runs that meet the 0.7 standard is 0.29. Therefore, approximately one-third of all 240 combinations of hypotheses that were modeled for action A1 meet the 24-year survival standard, while approximately two-thirds of the combinations do not meet the standard. Results for other actions and standards are summarized in Figure 2.2.4-2.

2.2.3 Relative Effects of Hypotheses on Outcomes

The results presented thus far provide useful preliminary information about the range of responses to management actions resulting from all combinations of hypotheses. However, they do not show which hypotheses exert the most influence over the results. This is important information because it focuses scientific debate and helps to prioritize further analytical efforts. For PATH, this information was also important for focussing the Weight of Evidence approach.

PATH used Categorical Regression Trees (CART) to identify these key hypotheses[6]. CART trees are a pictorial representation of the influence each of the hypotheses has in explaining the amount of variation in projected outcomes (a more detailed explanation of CART analyses can be found in Appendix E of the PATH Weight of Evidence Approach). Figures 2.2.3-1 to 2.2.3-4 are CART diagrams for the survival and recovery probabilities. Each branch on the tree represents a split between two or more alternative hypotheses that account for differences in the outcomes. The left side of each split represents hypotheses

[6] PATH conducted CART analyses on results for actions A1, A2, and A3 as part of the Weight of Evidence Process (see Appendix E of the *PATH Weight of Evidence Report*). The CART analyses shown here also include actions A2’ and B1. Results of these CART analyses are consistent with those conducted through the Weight of Evidence Process.

that result in lower survival and recovery probabilities; the right side represents hypotheses that produce higher probabilities. The most important hypotheses (those that account for the greatest amount of the differences in results) are split first (i.e., at the top), followed by progressively less important hypotheses at the bottom of the diagram. The vertical length of the branches are proportional to the amount of the variance explained by that hypothesis (i.e., longer branches have greater influence on the results). The mean value of the survival or recovery probability is shown at the end of each branch (i.e., at the bottom of the diagram).

To illustrate how to read the CART diagrams, consider Figure 2.2.3-1. The split between the extra mortality hypotheses indicates that the BKD and Regime Shift hypotheses result in significantly lower 24-year survival probabilities than the Hydro extra mortality hypothesis. Following the Hydro extra mortality branch, the next split is between the passage models, which indicates that within the Hydro extra mortality hypothesis, the FLUSH passage model results in significantly lower 24-year survival probabilities than the CRiSP model. Following the Hydro extra mortality / CRiSP branch further, the next split is between the life-cycle models, with the Alpha model producing lower jeopardy probabilities than the Delta model. Following the Hydro extra mortality / CRiSP / Delta model branch, the next split is between alternative hypotheses about predator removal effectiveness. Following the Hydro extra mortality / CRiSP / Delta / PREM3 branch further, the branch terminates in the average jeopardy probability for this particular combination of hypotheses (0.85). The absence of any further splits in this branch indicates that considering other hypotheses in addition to the combination of Hydro extra mortality / CRiSP / Delta / PREM3, would not make a significant contribution to the variation in results.

The results of these analyses show that the actions themselves are the most important influence in determining the 100-year survival and the recovery standards. A1, A2, and A2' tend to produce lower responses in these jeopardy probabilities, while A3 and B1 tend to produce higher responses. Interestingly, the actions generally account for a small proportion of the variance in the 24-year survival probabilities. For some branches, the effects of the actions are not significant at all. This means that in the short term, assumptions about how the system behaves have a greater effect than the choice of action in determining population levels. Of the alternative hypotheses, the extra mortality hypotheses, the passage and transportation assumptions, and the life-cycle models tend to account for the largest amount of the differences in results. Other hypotheses had minor effects on the outcomes.

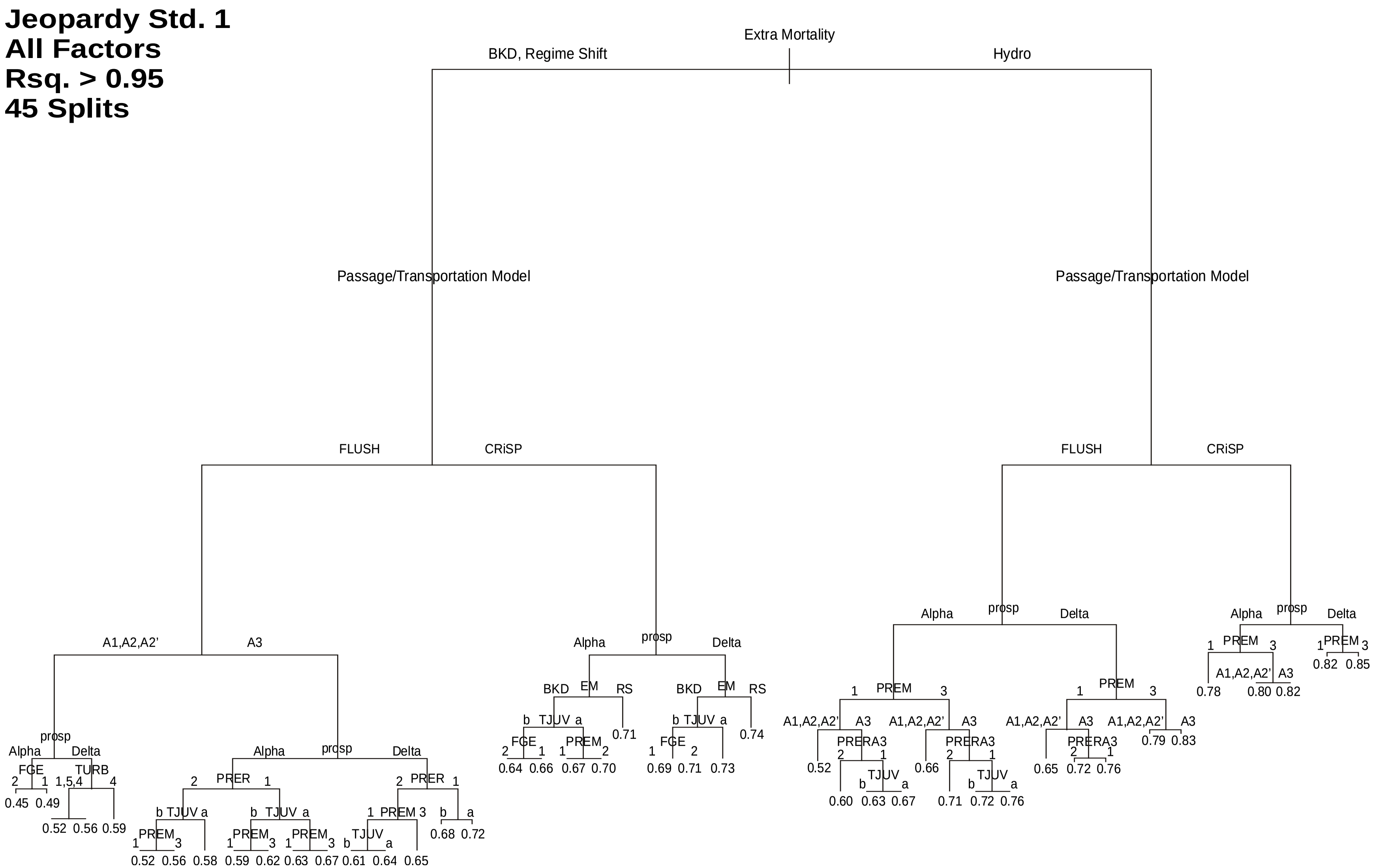


Figure 2.2.3-1: CART diagram for 24-year survival probabilities

Jeopardy Std. 2
All Factors
Rsq. > 0.95
19 Splits

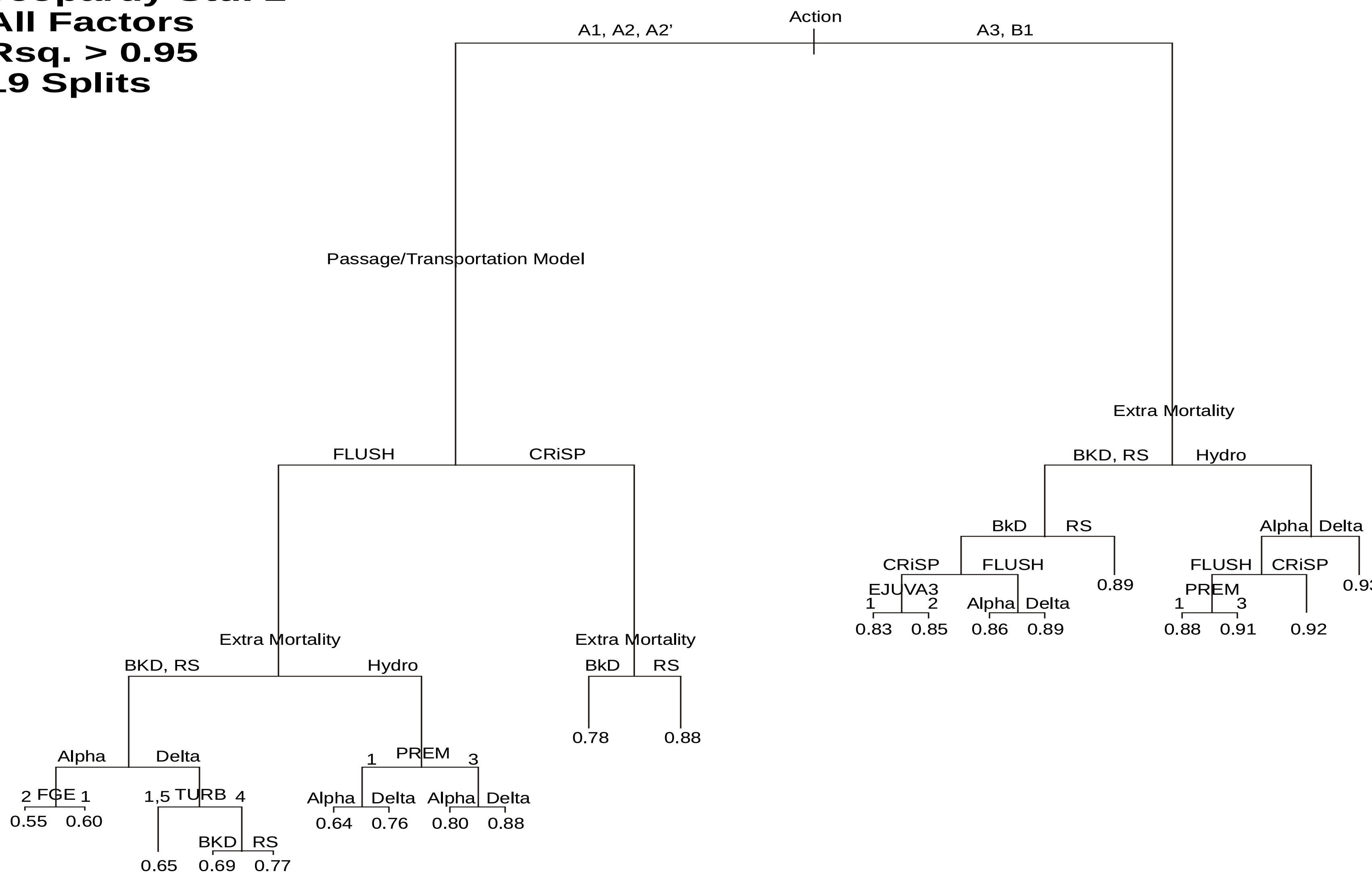


Figure 2.2.3-2: CART Diagram for 100-Year Survival Probabilities

Jeopardy Std. 3
All Factors
Rsq. > 0.95
45 Splits

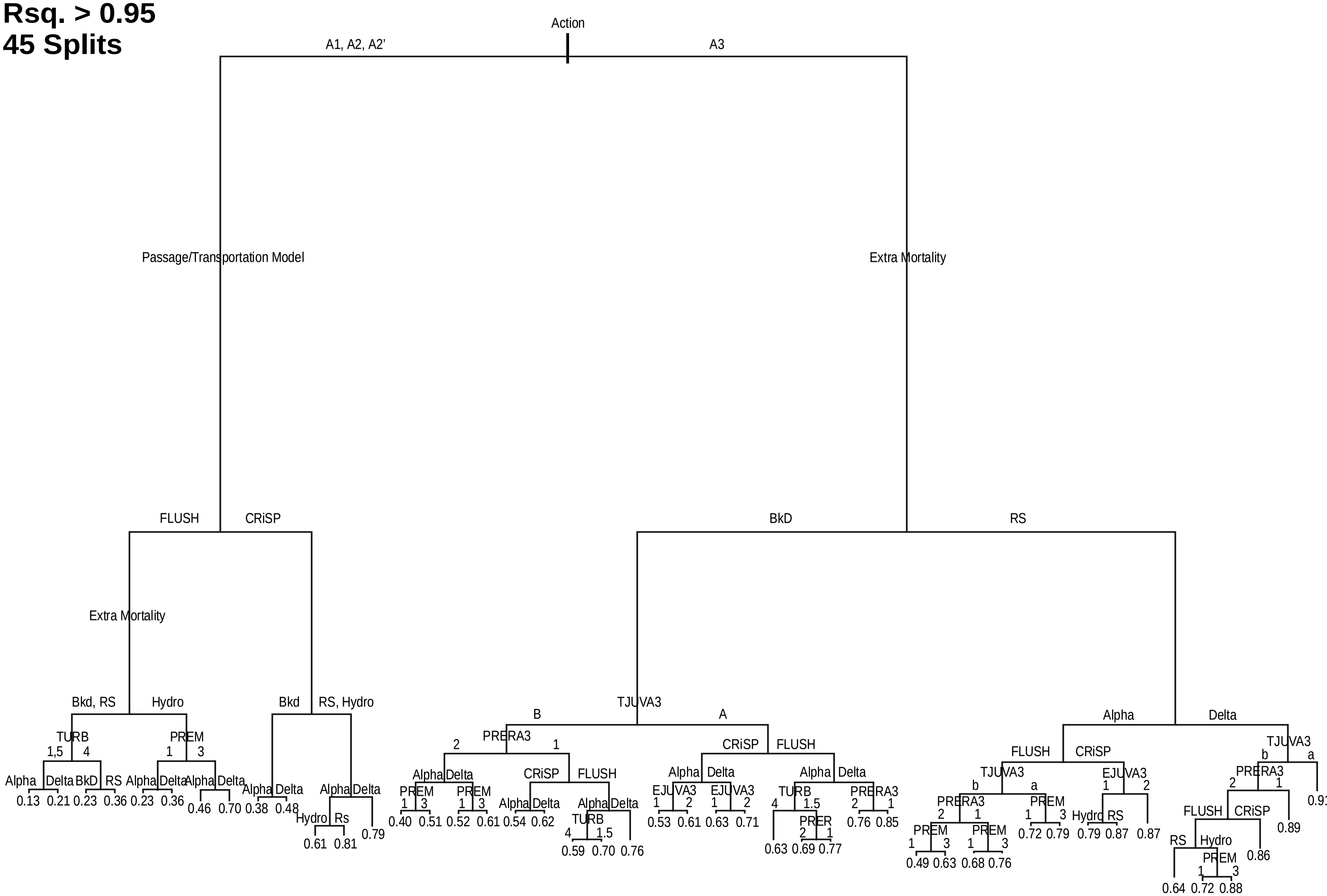


Figure 2.2.3-3: CART diagram for 24-year recovery probabilities

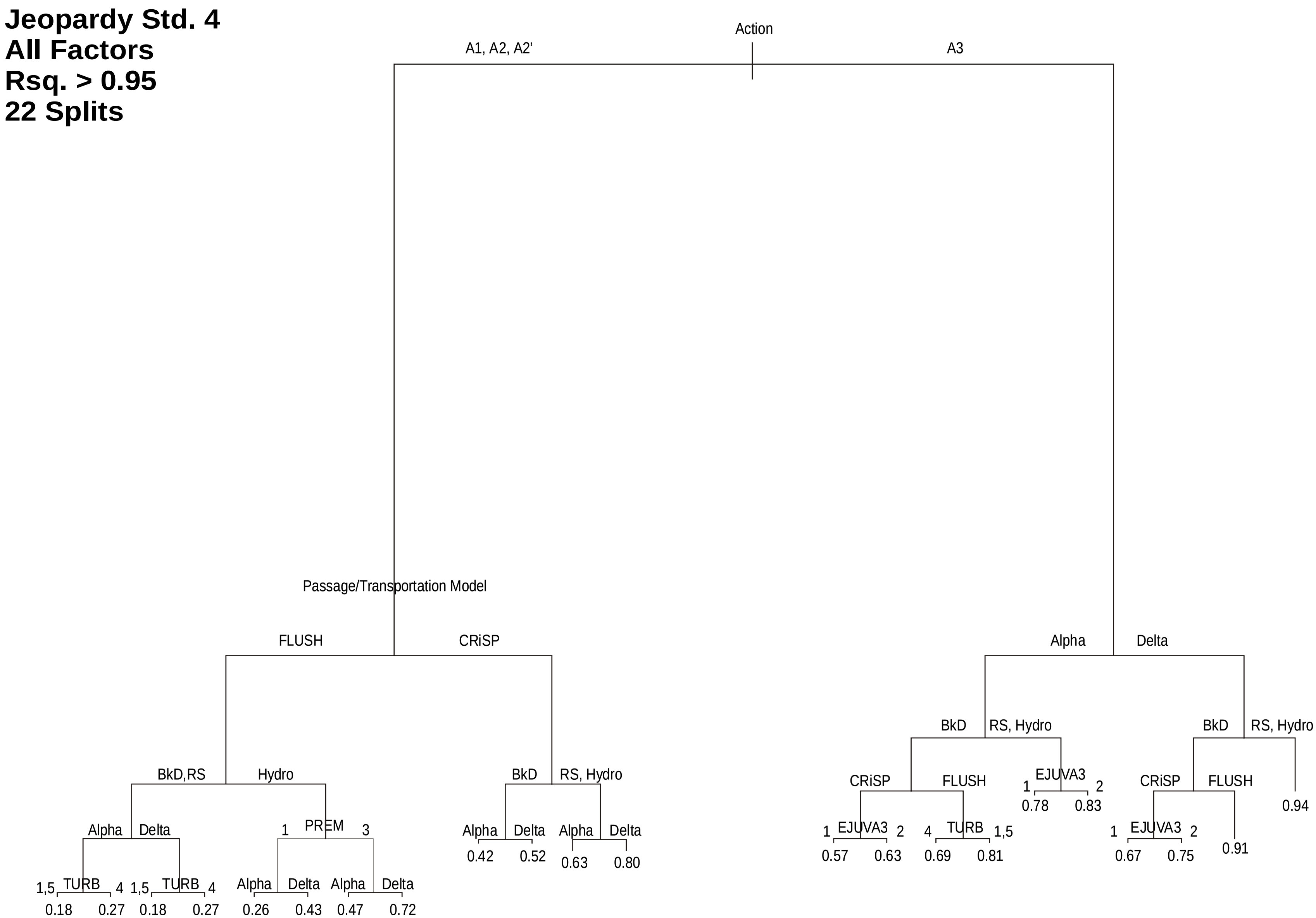


Figure 2.2.3-4: Cart diagram for 48-year recovery probabilities.

2.2.4 Weighted Results

The PATH Weight of Evidence Process

Results for spring/summer chinook are shown assuming both equal weights and applying the SRP weights derived through the Weight of Evidence Process. Assuming equal weights implies that all combinations of hypotheses are assumed to be equally likely. This is not necessarily an accurate assumption because there may be more or stronger evidence for some hypotheses than for others. Therefore, outcomes of alternative hypotheses that have more empirical support should be weighted more heavily than outcomes for hypotheses that have weaker support. Weighting the outcomes in this way provides a way to quantitatively incorporate the information available into decision-making.

Assigning these weights to alternative hypotheses was the objective of the PATH Weight of Evidence Process that was completed in FY98. Under that process, evidence for and against the most influential hypotheses (as determined by CART analyses) was compiled into the *PATH Weight of Evidence Report* and its associated submissions. This material was then reviewed by the PATH Scientific Review Panel (SRP). On the basis of that evidence, the SRP assigned relative weights to the alternative hypotheses. Documentation of the weighting process is contained in the PATH Scientific Review Panel's report *Conclusions and Recommendations from the PATH Weight of Evidence Workshop*; the weights themselves are summarized in Table 2.2.4-1.

Table 2.2.4-1: Weights obtained through Weight of Evidence Process.

Key Uncertainty	Alternative Hypothesis	Relative Weights			
		JC	SS	SC	CW
Passage/transportation Models	FLUSH	0.7	0.75	0.9	0.65
	CRiSP	0.3	0.25	0.1	0.35
Extra Mortality	BKD (here to stay)	0.3	0.25	0.495	0.4
	Hydro (here to stay unless dams go)	0.6	0.60	0.495	0.4
	Regime Shift	0.1	0.15	0.010	0.2
Life-cycle models	Alpha	0	0.7	0	0.1
	Delta	1	0.3	1	0.9
Length of transition period	2 years	0.6	0.33	0.2	0.5
	10 years	0.4	0.67	0.8	0.5
Historical turbine/bypass mortality	TURB 4 (Higher)	0.6	0.4	0.4	0.5
	TURB 5 (Lower)	0.4	0.6	0.6	0.5
Predator Removal Effectiveness	0%	0.7	1	0.8	0.9
	25%	0.3	0	0.2	0.1
Equilibrated Juvenile Survival Rate	0.85	0.6	0.8	0.8	0.25
	0.96	0.4	0.2	0.2	0.75

The weights that were obtained through the Weight of Evidence process were applied to the two summary statistics described in Section 2.2.2 to calculate a weighted average of the survival and recovery probabilities for a given action, and a weighted fraction of runs for a given action that met all of the jeopardy standards. The weighted results using these two summary statistics are summarized below. Some assumptions were necessary to apply the weights to the actions A2' and John Day drawdown. These actions had not yet been modeled when the Weight of Evidence Process was completed.

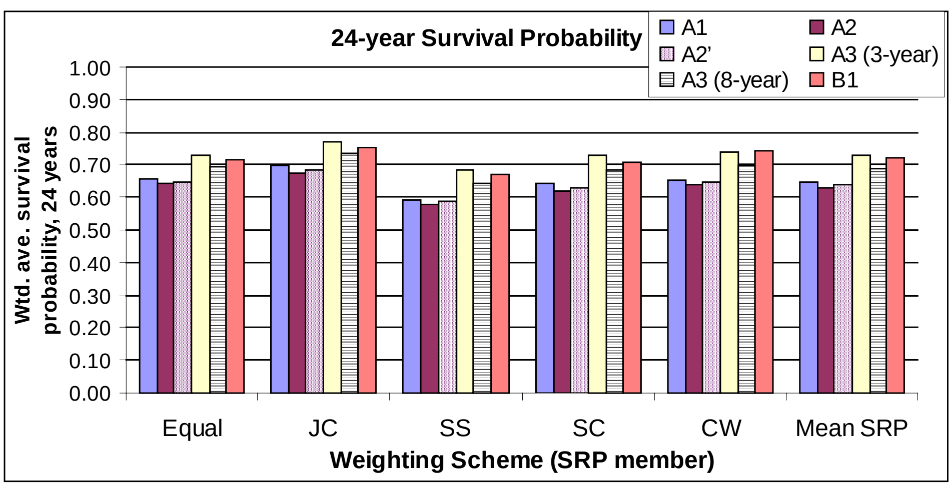
Consequently, alternative assumptions specific to these actions were not considered in the weight of evidence process. Appendix A describes these assumptions and their effects on the results in more detail.

Weighted Average Survival and Recovery Probabilities

Weights were applied to the survival and recovery probabilities to calculate a weighted average of these values (Figure 2.2.4-1). The procedure for doing this was to:

1. For each combination of hypotheses, calculate the overall weight to apply to that combination. The overall weight for any combinations is the product of the SRP's weights for each individual hypothesis within that combination. For example, for a combination of assumptions consisting of Assumption A, Assumption B, and Assumption C, the overall weight is the weight on Assumption A times the weight on Assumption B times the weight on Assumption C.
2. Multiply the jeopardy probability resulting from each combination of assumptions by the overall weight for the combination (as calculated in Step 1).
3. Sum the product calculated in Step 2 (i.e., jeopardy probability X weight) over all combinations for a particular action.

We show results using six weighting schemes: equal weights, each of the four SRP member's individual set of weights, and the mean of the four SRP member's individual weights. The graphs can be interpreted as follows: for A2 (for example), the average probability of exceeding the survival threshold over the first 24 years of the 100-year simulation period is 0.64, assuming that all combinations of hypotheses are equally likely. If the mean SRP weighting is applied, the weighted average is 0.63. Higher average jeopardy probabilities are better, because they suggest that there is more certainty in exceeding survival and recovery spawner thresholds.



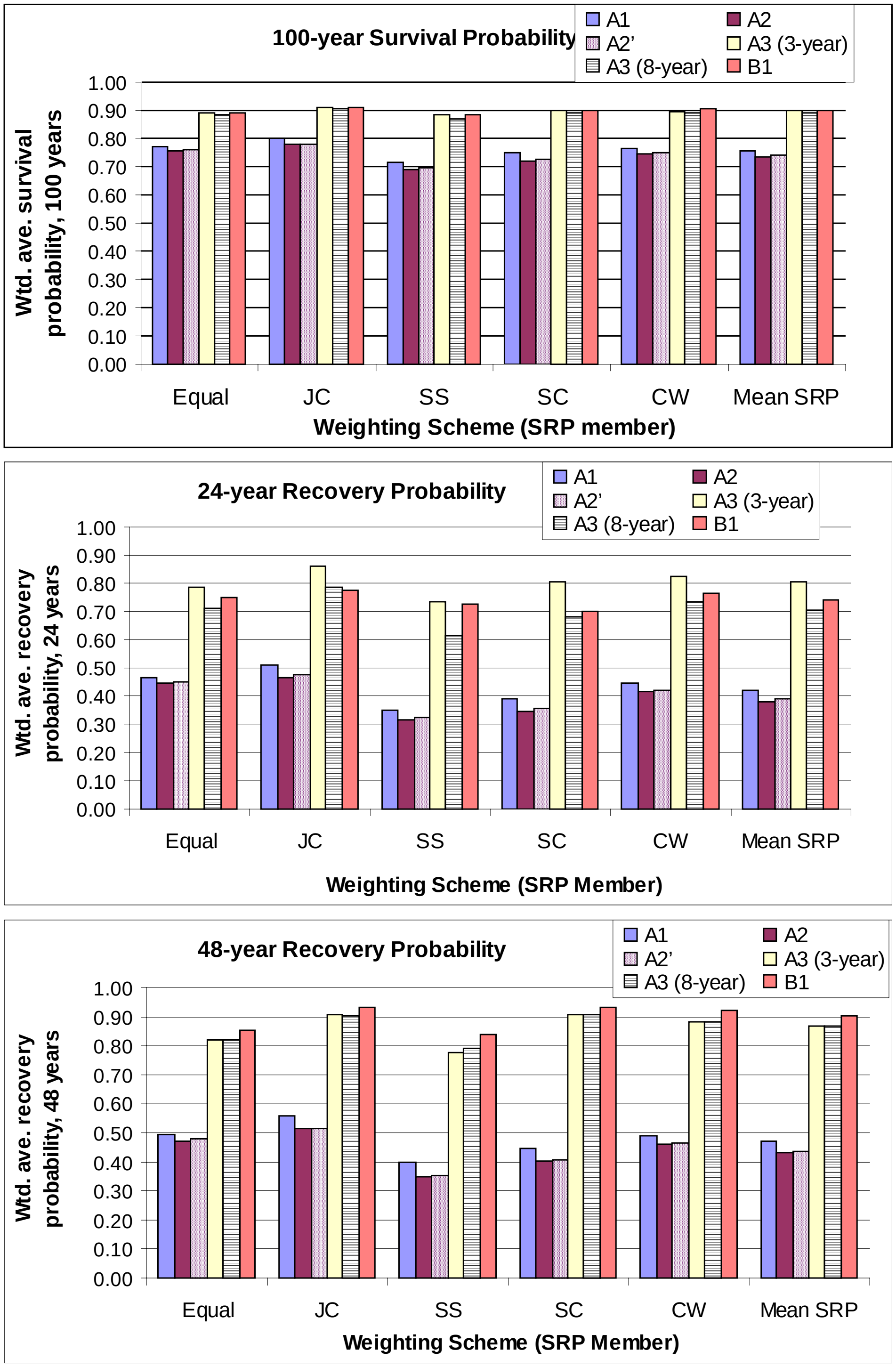


Figure 2.2.4-1: Weighted average survival and recovery probabilities for A1, A2, A2', A3 (with 3-year and 8-year delay), and B1. See Figure 2.2.2-1 and accompanying text for an explanation of how the averages were derived.

The weighted averages for the equal weight and the mean SRP weight cases are summarized in Table 2.2.4-2. In general, all of the non-drawdown actions (A1, A2, and A2') tend to produce similar results. Likewise, the drawdown actions (A3 and B1) also tend to produce results that are similar to each other. Average probabilities for the drawdown actions tend to be higher than the average results for the non-drawdown actions, particularly for the recovery probabilities.

Table 2.2.4-2: Average survival and recovery probabilities for spring summer chinook.

	24-year survival		100-year survival		24-year recovery		48-year recovery	
Action	Equal	Mean SRP	Equal	Mean SRP	Equal	Mean SRP	Equal	Mean SRP
A1	0.65	0.65	0.77	0.76	0.47	0.42	0.50	0.47
A2	0.64	0.63	0.76	0.73	0.44	0.38	0.47	0.43
A2'	0.65	0.64	0.76	0.74	0.45	0.39	0.48	0.44
A3 (3-year delay)	0.73	0.73	0.89	0.90	0.78	0.80	0.82	0.87
A3 (8-year delay)	0.69	0.69	0.88	0.89	0.71	0.70	0.82	0.87
B1	0.71	0.72	0.89	0.90	0.75	0.74	0.85	0.91

Weighted Fraction of Runs Meeting All of the Jeopardy Standards

The SRP weights were also used to calculate the weighted fraction of runs that met the jeopardy standards. Derivation of this fraction for action A1 and the 24-year survival standard was illustrated using Figure 2.2.2-1. Table 2.2.4-3 shows these results for the other actions and standards, using equal weights. With actions A1, A2, and A2', about 70% of the runs meet the 100-year survival standard, but only 42-47% meet the 48-year recovery standard and only 35-37% meet the 24-year survival standard. Under actions A3 and B1, 100% of the runs meet the NMFS 100-year survival and 48-year recovery standards, but a lower percentage of the runs meet the 24-year survival standard (47 to 63%).

Table 2.2.4-3: Weighted fraction of runs meeting survival and recovery standards.

Action	**24-year survival**	**100-year survival**	**48-year recovery**
A1	0.35	0.70	0.47
A2	0.35	0.68	0.42
A2'	0.37	0.69	0.45
A3 (3-year delay)	0.63	1.0	1.0
A3 (8-year delay)	0.47	1.0	1.0
B1	0.60	1.0	1.0

Results presented in Table 2.2.4-3 show the fraction of runs for a given action in which each individual jeopardy standard is met. However, it is important to know how well the actions meet **all** of the standards[7]. This is the most aggregated level of results, because it incorporates all uncertainties and all jeopardy standards into a single value that is relatively easy to understand. Management actions that have large values meet the standards under a broad range of possible hypotheses about future conditions (a robust action). Management alternatives with low fractions meet the standards under a narrower range of hypotheses.

7 i.e., meets the 24-year survival, the 100-year survival, and the 48-year recovery standard

These results are shown in Figure 2.2.4-2. The procedure for calculating the weighted fraction of meeting all of the standards was:

1. Calculate the overall weight for each combination of hypotheses (see Step 1 in the description of the calculation of weighted averages above).
2. For each combination of assumptions, look at the 24-year survival probability, the 100-year survival probability, and the 48-year recovery probability. All of the standards are met if the 24-year survival probability and the 100-year survival probability equal or exceed 0.7, and the 48-year recovery probability equals or exceed 0.5. If a combination of assumptions meets all of the standards, assign a score of 1, otherwise assign a score of 0.
3. Multiply the score (i.e., 1 if it meets all of the standards, 0 if it doesn't) for each combination of assumptions by the overall weight for the combination (as calculated in Step 1).
4. Sum the product calculated in Step 3 (i.e., score X weight) over all combinations for a particular action.

A3 with a three-year delay generally has the highest weighted fraction of runs meeting all of the standards, followed by B1, A3 with an eight-year delay, and A1, A2, and A2'. Although the exact order of the latter three depends on the weighting scheme, a general result is that the weighted fractions for the drawdown actions (A3 and B1) are higher than for non-drawdown actions (A1, A2, and A2'). Weighted fractions for the non-drawdown actions range from 0.2 to 0.4, depending on the weighting scheme applied. Weighted fractions for drawdown actions range from 0.4 to 0.8. Whether or not these fractions are "high enough" is a question we leave for policy-makers.

Note the similarity between the equally-weighted results in Figure 2.2.4-2 and the results for the 24-year survival standard in Table 2.2.4-3. This suggests that the 24-year survival standard is the most difficult to meet, and is the limiting factor in determining whether an action meets all of the standards for spring/summer chinook.

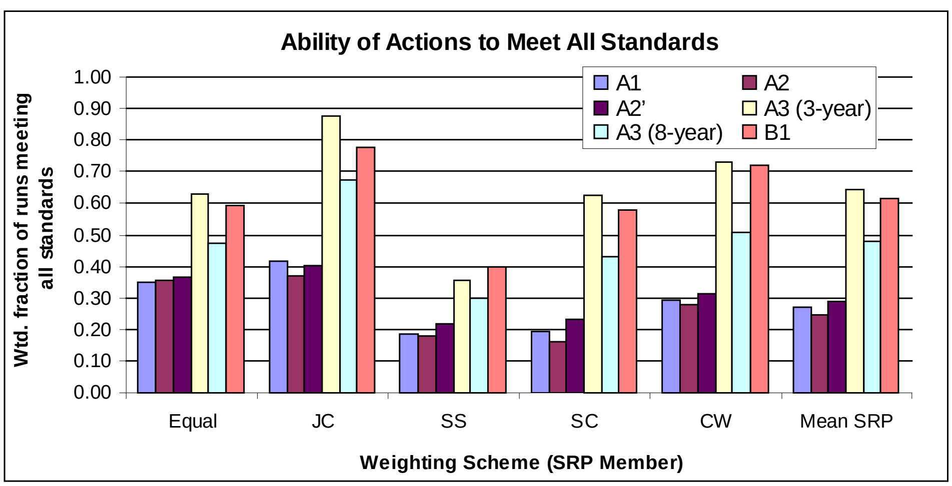


Figure 2.2.4-3: Weighted fraction of runs that meet all of the jeopardy standards (24-year and 100-year survival, and 48-year recovery).

Smolt to Adult Returns

Smolt to adult survival rates (SARs) estimate survival rates of fish from the time they pass the upper-most dam as smolts to the time they return to that dam as adults. SARs are estimated in the PATH prospective analyses by relating model estimates of survival over some historical time period to empirical measurements of SARs during that time period, and then projecting that relationship into the future. The PATH life-cycle model calculates a median SAR over the 100-year simulation period.

Previous analyses by the PATH hydro workgroup suggested that an SAR of between 2 and 6% be used as an interim goal for evaluating whether alternative actions meet survival and recovery standards (Chapter. 6 in *PATH FY1996 Retrospective Report*). This interim goal was based on Snake River and Warm Springs SARs during periods when those stocks were believed to be healthy, and on theoretical SARs associated with a range of Snake River egg-smolt survival rates from the last three decades.

A comparison of model-generated median SARs and jeopardy probability supports the idea that historical (pre-1970) SARs would be adequate for meeting the 100-year survival standard and 48-year recovery goals for spring/summer chinook, but may not be adequate for meeting the 24-year survival standard (see Figure 4.1-1). Median SARs must exceed 4% to achieve complete certainty of meeting the 48-year recovery standard, while meeting the 100-year survival standard requires a median SAR of at least 2%. A median of greater than 6% is needed to meet the 24-year survival standard with certainty.

All actions generate SARs that exceed historical (1977 to 1994) estimates (Table 2.2.4-4) Median SAR model estimates (to upper dam) were higher for action A3 than for actions A1 and A2. For A3, the median SAR was 4.0%, compared to 2.4% for A1 and 2.3% for A2. Minimum SARs for the three actions were 1.6%, 1.6% and 2.4% for A1, A2 and A3, respectively. Maximum SARs for the three actions were 4.9%, 4.8% and 7.1% for A1, A2 and A3, respectively. All actions appear to generate median SARs that are capable of meeting the 100-year survival standard (i.e., SAR > 2%), but only action A3 provides median SARs in the range to meet the 48-year recovery standard (SAR > 4%). None of the actions generate median SARs that would allow the 24-year survival standard to be met with certainty.

Table 2.2.4-4: Range and median SARs for actions A1, A2, and A3, and historical estimates for 1977 to 1994 (see Appendix B of this report).

	Minimum SAR	Median SAR	Maximum SAR
Historical (1977-1994)	0.2	1.0	2.6
A1	1.6	2.4	4.9
A2	1.6	2.3	4.8
A3	2.4	4.0	7.1

2.2.5 Qualitative Evaluation of A6/A6’

The IT directed PATH to evaluate the performance of an “inriver migration alternative” for the FCRPS. For spring/summer chinook, this alternative would assume:

- no transportation;
- functioning surface bypass systems and gas abatement structures (“major system improvements”) at all mainstem projects; and

- 1 Million Acre-Feet (MAF) for flow augmentation out of the upper Snake River basin above the 1995 Biological Opinion.

This hydrosystem alternative was designated "A6". The IT told PATH to pair this evaluation with A6', a version that removed all flow augmentation from the upper Snake basin, including the 427 kaf currently provided under the terms of NMFS' 1995 and supplemental FCRPS Biological Opinions.

The PATH Planning Group asked the authors of this report to delineate the information that the passage and life-cycle modelers would need to simulate the performance of the A6/A6' alternatives including:

- appropriate rules for the Corps' hydroregulation model runs;
- specific hardware improvements proposed for each of the FCRPS dams;
- expected fish passage efficiencies with those improvements in place; and
- expected changes in the spill program.

These parameters were discussed and will be provided for PATH review. However, the subgroup went further in developing a broad-brush analysis of the likely performance of A6 and A6' relative to A1, A2, and A3. The purpose of the analysis was to inform the IT regarding the expected performance of A6/A6' when PATH asks for guidance on the scheduling of tasks for FY99. For example, if A6/A6' would be likely to perform substantially worse than A2, the IT might advise NMFS and the Corps to drop A6/A6' from the Lower Snake Feasibility Study/EIS (FS/EIS). Alternatively, NMFS and the Corps might be instructed to evaluate these alternatives only in terms of the approximate relative biological effects (as in this report) and relative economic costs. On the other hand, if A6/A6' would be likely to perform much better than A2, the IT might direct PATH to move this task forward in these schedule to insure that more formal biological and economic analyses are included in the FS/EIS. This report describes the preliminary A6/A6' analysis, with sections on objectives, methods, results, discussion, and recommendations.

Objective

The objective of this preliminary analysis was to estimate the expected performance in terms of total system survival of A6/A6' relative to A1, A2, and A3, for spring/summer chinook. A similar analysis is possible for fall chinook but has not yet started pending instructions from the I.T.

The accuracy and precision of this preliminary analysis are very coarse compared to the passage/life cycle simulations that PATH has performed for A1, A2, and A3. Thus, the expected performance of A6/A6' is addressed only in relative terms. That is, would one of these alternatives perform:

- much worse than;
- approximately the same as; or
- much better than A2.

It is not possible, without substantial additional analysis, to describe the performance of A6/A6' on a finer scale. The scenarios described in this report capture a range of assumptions about potential increases in the proportion of fish that would pass FCRPS projects by non-turbine routes given "major system improvements" (surface bypass) and potential increases in reservoir survival due to an additional 1 MAF out of the upper Snake. These combinations of assumptions may or may not be realistic and do not include the effects of gas abatement modifications, for example. The question of the accuracy of the assumptions regarding the effectiveness of surface bypass and gas abatement would best be answered by a collaborative examination of the data sets developed in prototype testing of the structures that would be implemented.

Methods

Per-Project Survival Improvements

In general, improvements in per-project (1) dam passage survival and (2) reservoir survival, due to A6/A6' measures, were calculated as a function of the corresponding survival rates observed under the base-case, A2:

$$S_{improved(i,j)} = S_{base(i,j)} + \left(\left[1 - S_{base(i,j)}\right] \times \left[\text{Proportional Improvement}\right]\right) \qquad \text{[Eqn. 1]}$$

where:

$S_{improved(i,j)}$	=	improved FPE or reservoir survival at project *i* in year *j*
S_{base}	=	survival at project *i* in year *j* under A2 (base case), bounded between 0 and 1
Proportional Improvement	=	improvement in FPE or reservoir survival, expressed as proportion of the value observed under the base case

Survival improvements are proportional to existing survival rates (i.e., the rates predicted by FLUSH and CRiSP for the base case, A2). To keep improvements from exceeding unity, they were applied as reductions in mortality rates. For example, an improvement in reservoir passage survival of 50% (0.5), applied to a base rate (from FLUSH or CRiSP) of 0.6, would increase the reservoir survival rate to 0.8:

$$0.6 + \left[(1 - 0.6) \times 0.5\right] = 0.8 \qquad \text{[Eqn. 2]}$$

Similarly, a passage survival improvement of 0.5 would increase a base-case survival rate of 0.9 to 0.95. Dam passage survival was bounded on the upper end at 0.98, which is equivalent to stating that zero fish would go through the turbines and that the passage survival of those fish routed via the juvenile bypass and spillway would be 98%.

The improved survival at a given reservoir *(i)* in a given year *(j)* was calculated using Eqn 1 directly:

$$S_{res,improved(i,j)} = S_{res,base(i,j)} + \left(\left[1 - S_{res,base(i,j)}\right] \times \left[\text{Proportional Improvement}\right]\right) \qquad \text{[Eqn. 3]}$$

where:

$S_{res,\ base(i,j)}$	=	base-case reservoir survival per project (*i = 1-8*), per year (*j = 1-14* for 1977 through 1990) under the base case (A2)
Improvement	=	proportional improvement
$S_{res,\ improved(i,j)}$	=	improved reservoir survival for project *i* during year *j*

The "base" values for reservoir survival were obtained from FLUSH and CRiSP prospective passage model output on the basis of "per mainstem project" (*i = 1-8*), "per year" (*j = 1-14* for the outmigration years 1977 through 1990).

Because dam passage survival is a function of bypass survival and turbine survival, the calculation for this parameter was slightly more complex. In order to simulate the effect of an increase in FPE due to surface bypass systems, it was necessary to extract FPE from each of the FLUSH and CRiSP dam passage survival estimates (assuming FGE1) per project, per year. Fish passage efficiency was extracted by rearranging the following equation:

$$S_{dam,\,base} = FPE_{base} \times (\text{Bypass or spill survival rate})_{base} + [1 - FPE_{base} \times (\text{Turbine survival rate})_{base}] \quad \text{[Eqn. 4]}$$

where:

FPE_{base} = fish passage efficiency (the proportion of fish passing a project by non-turbine routes [spill plus bypass]) under the base case (A2)

To simulate the effect of adding a surface bypass system at each project, an improvement in FPE was calculated, using Eqn 5:

$$FPE_{improvedIi,\,j)} = FPE_{base(i,\,j)} + [1 - FPE_{base(i,\,j)}] \times \text{Proportional Improvement} \quad \text{[Eqn. 5]}$$

where:

$FPE_{base(i,j)}$ = FPE for the base case (A2) at project *i* during year *j*

$FPE_{improved(i,j)}$ = improved FPE at project *i* during year *j*

Then, the improved FPE was used to calculate improved dam survival (at project *i* during year *j*) as:

$$S_{dam,\,improved\,(i,\,j)} = (FPE_{improved} \times [\text{Bypass or spillway survival rate}]) + ([1 - FPE_{improved}] \times (\text{Bypass or spillway survival rate}))$$

[Eqn. 6]

Improved project survival (i.e., for a specific dam and reservoir combined) in a specific year is:

$$S_{improved\,(i,\,j)} = S_{dam,\,improved\,(i,\,j)} \times S_{res,\,improved\,(i,\,j)} \quad \text{[Eqn. 7]}$$

Base-Case Parameters

Dam Passage Survival

We assumed the base-case (A2) passage routing efficiencies and survival rates shown in Table 2.2.5-1.

Table 2.2.5-1: Routing efficiency and survival assumptions under the base case (A2).

Route	Routing Efficiency	Survival[8]
Turbines	1 – [FPE]	0.90
Bypass + Spillway	FPE	0.98

8 The survival rates in Table 2.2.5-1 are similar to those used in the passage model assumption TURB1 (Marmorek and Peters 1998).

As a simplifying assumption, the base-case survival rates of fish routed through surface bypass systems were assumed to be equal to survival rates for fish routed over the spillway. This assumption is based on the results of prototype surface bypass tests at Lower Granite and Bonneville dams (Johnson et al. 1998; Adams and Rondorf 1998).

Reservoir Survival

The hydroregulation model output ("hydroregs") used in the passage models to simulate the base case (A2) assumed that 427 kaf would be available to augment flows to meet NMFS biological opinion flow objectives. Alternative A6 assumes the availability of an additional 1 MAF (total of 1.427 MAF). Alternative A6' assumes that no water comes out of the upper Snake basin for flow augmentation, removing even the 427 kaf used in A2.

If the IT requests that PATH perform full passage/life-cycle modeling for A6, PATH would need more guidance regarding the timing when the additional 1 MAF would be applied under A6. In the existing Corps hydroreg model for A6, spring flow objectives are prioritized over refill of storage reservoirs by June 30th. However, that hydroreg model output was not used in this simple modeling effort. Instead, a larger volume, 2 MAF, was applied evenly across the juvenile outmigration season (mid-April through August). Earl Weber performed this analysis for one of the CRITFC tribes, simulating inriver survival for 1977 through 1992 using FLUSH. In this scenario, the FLUSH model predicted that (eight-project) inriver system survival increased roughly 4 %[9]. This translates to a proportional improvement of 0.14, the value used to simulate the "most realistic" improvement in per-project survival under A6. In this broad-brush analysis, removal of all flow augmentation from the upper Snake basin was assumed to result in a decrease in reservoir survival half the size of the positive improvement due to an additional 2 MAF (i.e., -0.07).

System Survival Improvements

The improved ***per-project*** survival rates $S_{improved(i,j)}$ were averaged over all years (i.e., $j = 1\text{-}14$) and improved ***system*** survival was calculated by raising the average annual improved per project survival to the eighth power.

$$S_{sys,\,improved} = \left[avg\ S_{improved\,(i)}\right]^\wedge 8 \times \left(\left[D \times P\right] + 1 - P\right) \qquad \text{[Eqn. 8]}$$

where:

$S_{sys,\ improved}$	=	total improved system survival through the hydrosystem: the number of inriver-equivalent smolts below Bonneville Dam divided by the population at the head of Lower Granite reservoir (includes post-Bonneville mortality for transported fish) (averaged over years $j = 1\text{-}14$)
avg $S_{improved(i)}$	=	improved per-project (i) survival (averaged over the years $j = 1\text{-}14$)
D	=	ratio of post-Bonneville survival for transported fish to that for non-transported fish
P	=	proportion of fish below Bonneville that were transported

[9] If run through the CRiSP passage model for spring/summer chinook, a flow augmentation volume of 2 MAF would probably result in a survival improvement substantially less than that predicted by FLUSH. CRiSP predicts a weaker association between flow and reservoir survival.

Under A6/A6’, the “proportion of fish below Bonneville that were transported” would be zero[3]. Because “P” in Eqn. 8 is zero, system survival would be that of inriver migrants. However, under A2, a high percentage of smolts are transported, and the equation above was used to account for the post-Bonneville mortality of transported fish relative to that of inriver migrants.

Data Sources

The base-case dam and reservoir survival estimates in Tables 2.2.5-2 and 2.2.5-3, respectively**,** were taken from the FLUSH and CRiSP diagnostic output for A2. These were the only outputs generated with separate dam and reservoir passage survivals. The values for “P” were taken from the FLUSH and CRiSP prospective model passage output files. The FLUSH “D” values were also taken from the prospective passage output; the CRiSP “D” values were updated per an agreement among the modelers. Both the FLUSH and CRiSP runs for A2 used the FGE1 assumption (i.e., bypass improvements result in an improvement in FGE).

Table 2.2.5-2: Mean per project dam survival and FPE under the base case (A2).

	Mean Per Project Survival
Per Project Dam Survival	
FLUSH, FGE1	0.96
CRiSP, FGE1	0.96
FPE[10]	
FLUSH, FGE1	0.78
CRiSP, FGE1	0.80

Table 2.2.5-3: Per-project reservoir survival under PREM1 and PREM3[11] for FLUSH and CRiSP under the base case (A2).

	Mean Per Reservoir Survival
FLUSH	
FGE1,PREM1	0.83
FGE1, PREM3	0.87
CRiSP	
FGE1, PREM1	0.93
FGE1, PREM3	0.95

Scenarios and Comparison to the Base-Case

We evaluated a number of scenarios, pairing hypothetical improvements in FPE (due to implementation of surface bypass systems) with improvements or declines in reservoir survival with 1.427 MAF or no flow augmentation (Table 2.2.5-4).

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10 FPE is derived from Equation 5, where FPE_{base} is calculated from the S_{dam} estimates in the prospective passage model diagnostics and turbine and bypass survivals are those shown in Table 2.2.5-1.

[11] The two hypotheses regarding the effect of predator removal were treated as sensitivities. PREM1 assumes that predator removal has not effect; PREM3 assumes that predator removal improves inriver survival.

Table 2.2.5-4: Estimates of project and total system survival under various combinations of proportional improvements in FPE and reservoir passage survival.

	Proportional Improvement		**Passage Survival**		**S_{system}**
Scenario	**Dam Passage Efficiency (FPEO**	**Reservoir Passage**	$S_{project}$	**Total S_{system}**	**Ratio A6:A2**
FLUSH, FGE1, PREM1	0.50	0.00	0.80	0.17	**0.66**
FLUSH, FGE1, PREM3			0.85	0.27	**0.65**
CRiSP, FGE1, PREM1			0.91	0.48	**0.91**
CRiSP, FGE1, PREM3			0.93	0.55	**0.95**
FLUSH, FGE1, PREM1	0.00	0.50	0.88	0.36	**1.35**
FLUSH, FGE1, PREM3			0.90	0.44	**1.05**
CRiSP, FGE1, PREM1			0.94	0.63	**1.21**
CRiSP, FGE1, PREM3			0.95	0.67	**1.16**
FLUSH, FGE1, PREM1	0.50	0.50	0.89	0.39	**1.45**
FLUSH, FGE1, PREM3			0.91	0.47	**1.13**
CRiSP, FGE1, PREM1			0.94	0.64	**1.22**
CRiSP, FGE1, PREM3			0.95	0.68	**1.18**
FLUSH, FGE1, PREM1	1.00	0.00	0.81	0.19	**0.71**
FLUSH, FGE1, PREM3			0.86	0.29	**0.70**
CRiSP, FGE1, PREM1			0.91	0.48	**0.92**
CRiSP, FGE1, PREM3			0.93	0.55	**0.96**
FLUSH, FGE1, PREM1	1.00	0.10	0.83	0.22	**0.83**
FLUSH, FGE1, PREM3			0.87	0.32	**0.78**
CRiSP, FGE1, PREM1			0.92	0.51	**0.98**
CRiSP, FGE1, PREM3			0.93	0.58	**1.00**
FLUSH, FGE1, PREM1	0.83	0.140[12]	0.83	0.23	**0.87**
FLUSH, FGE1, PREM3			0.87	0.33	**0.80**
CRiSP, FGE1, PREM1			0.92	0.52	**1.00**
CRiSP, FGE1, PREM3			0.94	0.59	**1.01**
FLUSH, FGE1, PREM1	0.83	0.000	0.81	0.18	**0.69**
FLUSH, FGE1, PREM3			0.85	0.28	**0.68**
CRiSP, FGE1, PREM1			0.91	0.48	**0.92**
CRiSP, FGE1, PREM3			0.93	0.55	**0.95**
FLUSH, FGE1, PREM1	**0.83**	**-0.070[13]**	**0.80**	**0.16**	0.68
FLUSH, FGE1, PREM3			**0.84**	**0.26**	0.63
CRiSP, FGE1, PREM1			**0.91**	**0.46**	0.88
CRiSP, FGE1, PREM3			**0.92**	**0.53**	0.92

For each group of four scenarios, base-case estimates of total system survival (used to compute the ratio of A6 to A2 [Table 2.2.5-4]) were taken from the passage model runs produced for diagnostics (Table 2.2.5-5). Per-project survival is calculated as the eighth root of system survival.

[12]Equivalent to approximately a 5% overall increase in reservoir survival, per Earl Weber's analysis, intended to simulate the addition of 1 MAF flow augmentation (i.e., A6).

[13]Equivalent to approximately a 2.5% overall decrease in reservoir survival to simulate the reduction of baseline flow augmentation by 427 kaf (i.e., A6').

Table 2.2.5-5: Per-project and system survivals[14] under hydrosystem alternative A2, best and worse case passage model runs.

Passage Model	Rank	Per Project Survival	System Survival
FLUSH	Worst	0.85	0.27
	Best	0.90	0.41
CRiSP	Worst	0.92	0.52
	Best	0.93	0.58

For example, the FLUSH "worst" system survival estimate in Table 2.2.5-5 is the value of A2 to which the FLUSH estimate for A6 (assuming PREM1) is compared (0.17 / 0.27 = 0.66, allowing for spreadsheet rounding). The FLUSH "best" system survival estimate is the value for A2 to which the FLUSH estimate for A6 (PREM3) is compared (0.27 / 0.41 = 0.65).

Results

Results are shown in the last three columns of Table 2.2.5-4. For each set of four FLUSH and CRiSP simulations, the most consistently high ratios of A6:A2 occur when an FPE improvement of 0.50 was paired with the same level of reservoir survival improvement (i.e., 0.50). Increasing the proportional reservoir survival component (presumably due to the additional 1 MAF from the upper Snake) from 0.00 to 0.50 would double the ratio of A6:A2 under FLUSH and would increase the ratio by approximately one third under CRiSP.

The third-to-last scenario in Table 2.2.5-4 contains the A6/A6' workgroup's best guess regarding realistic values for each variable (per-project FPE improvement = 0.83[15]; per-project reservoir survival improvement = 0.14)[16]. Chris Pinney ventured an expectation, based on prototype testing at Lower Granite and Bonneville dams, that full powerhouse surface bypass, when combined with extended screens, could increase FPE to about 0.95 (based on per-project survival improvement of 0.83. The per-project proportional improvement in reservoir survival was derived from Earl Weber's 2 MAF analysis, described in Section 2.2.2. The second-to-last scenario is similar, but assumes no increase in reservoir survival. The difference in system survival for these two scenarios is about 4% for each of the four cases, roughly the same as that derived by Earl using FLUSH for the 2 MAF increase.

The last scenario in Table 2.2.5-4 incorporates a negative per-reservoir survival improvement [minus 0.07], to simulate an effect of removing the existing 427 kaf from the upper Snake. Absent any field data, a value of half the magnitude, and opposite in sign, to the effect of adding the extra 1 MAF was chosen. As shown in Table 2.2.5-4, under these conditions, total system survival of spring/summer chinook with no flow augmentation from the upper Snake would be about 6-7% less than survival with 1.427 MAF. However, our mini-simulation only examines effects within the eight-project federal hydrosystem. Effects (if any) of flow augmentation on the survival of spring/summer chinook in the reach above Lower Granite pool or below Bonneville tailrace are not addressed.

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14 System survivals taken from the passage model diagnostics include post-Bonneville mortality for transported fish.

[15] A per-project improvement in FPE of 0.83 creates an upper bound on the absolute value of per-project FPE equal to 95%.

[16] According to Earl Weber's analysis, a proportional improvement of 0.14 is needed to increase overall reservoir survival by 4%.

Discussion

Using CRiSP passage model output as the baseline, A6 would be likely to perform about as well as A2. When estimates are based on FLUSH output, A6 performed much worse that A2 except when reservoir survival was assumed to increase 50%, enough to compensate for the loss of direct survival benefits achieved through transportation.

Scenario 6 in Table 2.2.5-4 (considered more realistic than the others) would result in fractional per-project improvements in FPE of no more than 0.83 and in reservoir survival of no more than 0.14. The estimate for FPE is based on results of field tests at Lower Granite and Bonneville dams. The estimate of improvement in reservoir survival is based on Earl Weber's analysis, described in Section 2.2.2. In this "most realistic" scenario, A6 would be likely to perform about as well as A2 under CRiSP and worse than A2 under FLUSH.

We assumed that removing the 427 kaf would result in a decrease in reservoir survival half the size of the positive improvement due to an additional 2 MAF. This translated to a per-project improvement fraction of –0.07. Under both passage models, this scenario performs worse than A2.

Recommendations

Based on these findings, the PATH recommends that A6/A6' be given a low priority at this time for further spring/summer chinook passage/life cycle modeling.

Literature Cited and Other References

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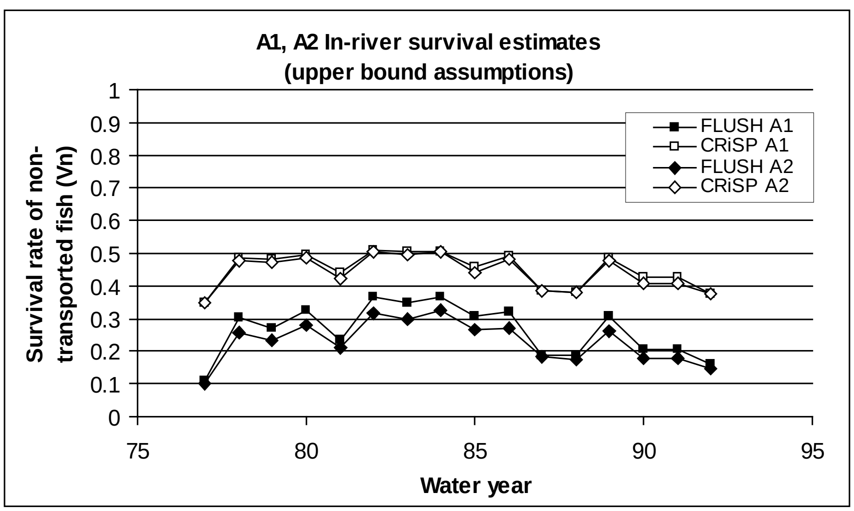
2.2.6 Comparison of Passage Survival Measures

The regression trees shown in Section 2.2.3 clearly show the large relative influence of CRiSP and FLUSH passage models and their associated transportation models on the outcomes of the actions. Previous diagnostic analyses (Section A.2.1 of the *Preliminary Decision Analysis Report*, Appendix D of the *PATH Weight of Evidence Report*) have identified the specific components of the two passage/transportation models that account for the majority of the differences in their outputs. The in-river survival estimates produced by the two models are particularly important because not only do they determine the direct survival of in-river fish, they also are a key determinant in survival outside of the hydrosystem through their influence on D values (the ratio of post-Bonneville survival of transported and non-transported fish) and on extra mortality.

In this section, we simply compare passage model estimates of various measures of survival through the migration corridor. Rather than repeating detailed diagnostic analyses of these measures, the objective of this section is to simply lay out these estimates for all actions side-by-side as an aid to understanding the overall results of the PATH analyses.

In-river Survival Rate

In-river survival is defined as the survival rate of non-transported fish from the head of Lower Granite pool to below Bonneville Dam. FLUSH and CRiSP estimates of these values are shown in Figure 2.2.6-1, for an upper and lower bound. Results for A2' are not shown because they are virtually identical to results for A2. The upper bound consists of optimistic assumptions about FGE and the effectiveness of the predator removal program, while the lower bound consists of pessimistic assumptions about these hypotheses. For the drawdown runs, the upper bound also includes an optimistic hypothesis about the equilibrated juvenile survival rate, which is based on pre-dam survival rate estimates through the relevant reaches. The lower bound for drawdown runs includes a more pessimistic hypothesis about the equilibrated juvenile survival rate, which is based on current estimates of survival of spring/summer chinook through free-flowing reaches of the Snake River. Further sensitivity analyses of these assumptions for juvenile survival rates following drawdown have been proposed for FY99.



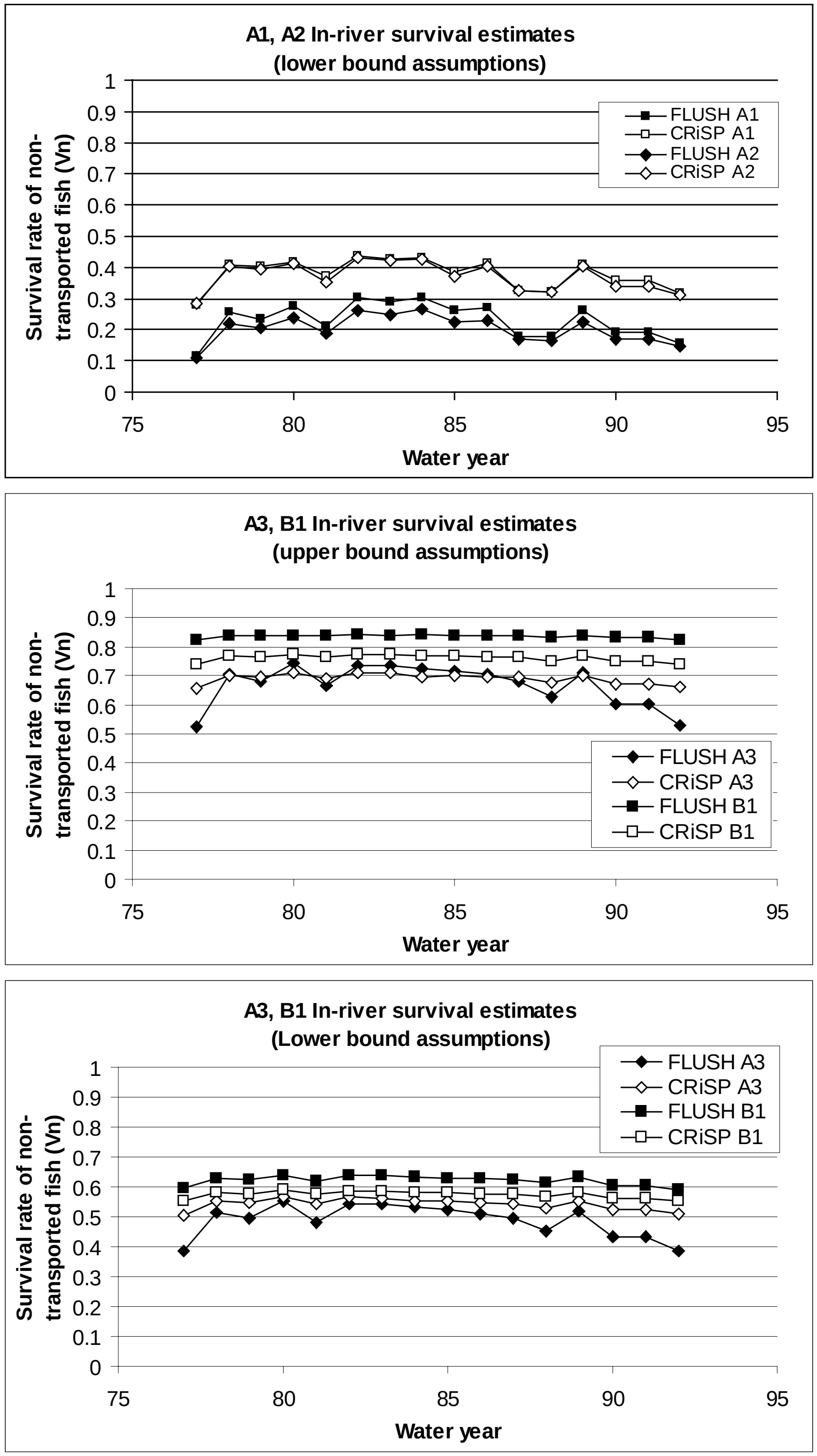


Figure 2.2.6-1: CRiSP and FLUSH estimates of in-river survival of non-transported fish from the head of Lower Granite pool to below Bonneville for actions A1, A2, A3, and B1.

Total Survival Rate to below Bonneville

For non-drawdown actions (A1, A2, and A2'), the vast majority of fish are transported, and the in-river survival rate of non-transported fish shown above applies only to a small fraction of the fish that survival to below Bonneville. Therefore, we also show the overall survival of transported and non-transported fish from the head of Lower Granite pool to below Bonneville for these actions, again for an upper and lower bound set of FGE and predator removal assumptions (exp(-M); Figure 2.2.6-2). The total survival rate for drawdown actions is the same as the in-river survival rate (Figure 2.2.6-1), because none of the fish are transported in these actions. Since the overall survival rate is closely related to how many fish were transported, we also show the proportion of fish below Bonneville that were transported (Pbt) (Figure 2.2.6-3; we show only the upper bound for Pbt; the lower bound is similar).

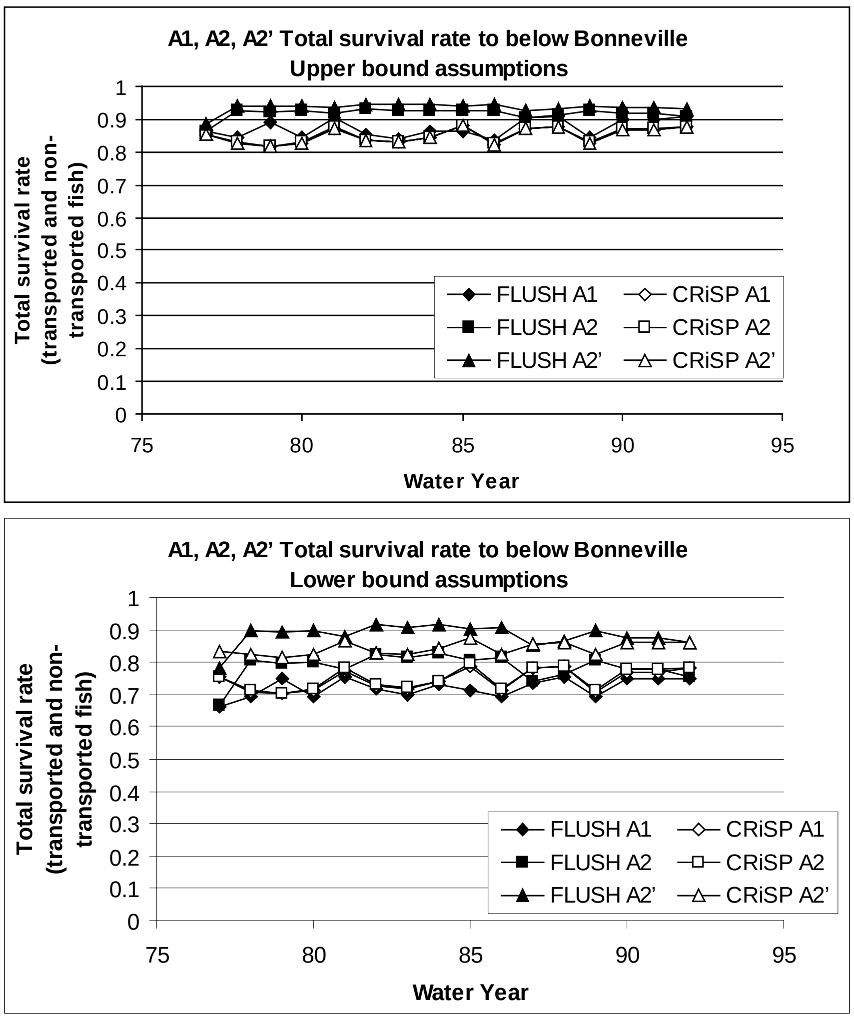


Figure 2.2.6-2: FLUSH and CRiSP estimates of total overall survival rate of transported and non-transported fish, from head of Lower Granite pool to below Bonneville, for actions A1, A2, and A2'. CRiSP A1 estimates are virtually identical to CRiSP A2 estimates.

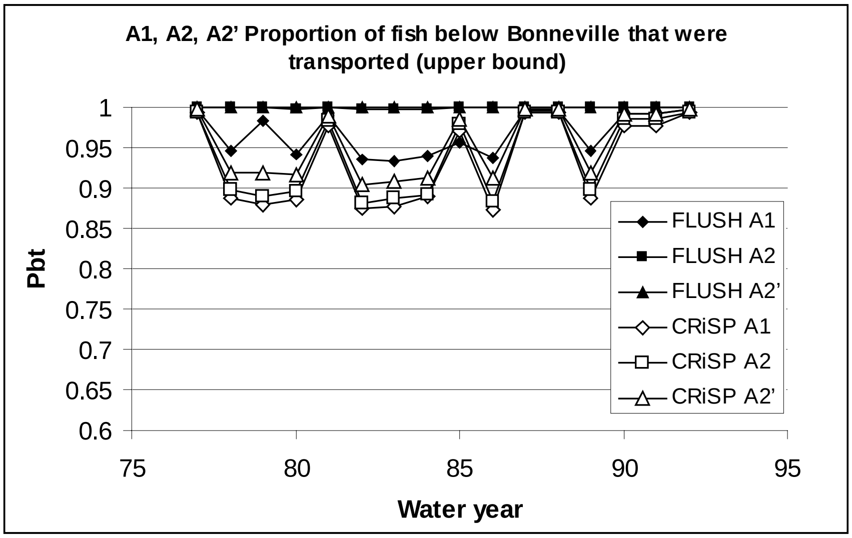


Figure 2.2.6-3: CRiSP and FLUSH estimates of proportion of fish below Bonneville that were transported for actions A1, A2, and A2'.

2.3 Sensitivity Analyses for Spring/Summer Chinook

In addition to the uncertainties explored formally through the decision analysis (see Section 2.2.1), PATH has also explored the effects of other uncertainties and assumptions through less formal sensitivity analyses.

The PATH *Weight of Evidence Report* contained sensitivity analyses of the following assumptions:

Alternative transportation model and assumptions used in conjunction with the FLUSH passage model (Appendix C and I of the WOE Report).

Further sensitivity and diagnostic analyses of the components of the passage and transportation models (Appendix D).

Issues related to the implementation of the transportation models in CRiSP (Appendix F).

Sensitivity analyses of the Delta model to various components of the spawner-recruit data (Appendix G).

An alternative formulation of the Hydro extra mortality hypothesis (Appendix H).

Sensitivity of A3 results to assumptions about the juvenile survival rate during the transition period (Section 5.4 of the Weight of Evidence Report)

In this section of the FY98 Final Report, we look at the effects of four additional assumptions:

1. Effects of alternative habitat scenarios and implementation of scenarios.
2. Effects of alternative harvest rate schedules.
3. Effects of explicitly incorporating sources of mortality that are not captured in the historical data (e.g., mortality due to bird predation).
4. Increasing adult conversion rates with John Day drawdown

These sensitivity analyses involved doing a small number of runs with variations on certain hypotheses, then comparing the jeopardy probabilities[17] from these sensitivity runs with the baseline cases. Further exploration of the effects of uncertainties which appear to have large influences on these results is planned for FY99.

2.3.1 Effects of Alternative Habitat Scenarios and Implementation of Scenarios

Although the primary focus of the current PATH prospective analyses is on the evaluation of hydrosystem actions, PATH has also explored the sensitivity of responses to hydrosystem actions to three alternative habitat protection and restoration scenarios[18]. These scenarios were developed by the PATH habitat sub-group using their collective professional judgement about the likely effects of future habitat scenarios on stock productivity, and are documented in Section A.3.5 of the *PATH Preliminary Decision Analysis Report*.

Scenario “0”: base case – future habitat scenarios have no effect on stock productivity
Scenario “A”: continued management according to existing habitat management plans
Scenario “B”: all practical measures taken to protect and restore fish habitat

Scenarios A and B are defined by probabilities of an increase, decrease, or no change in each Snake River index stock’s productivity parameter (the Ricker a parameter) over 12, 24, and 48-year time periods. These probabilities reflect current habitat conditions as well as possible effects of future habitat scenarios. This is an important point, because it means that the productivity of stocks that are:

a) currently experiencing gradual or periodic declines in habitat quality may continue to decline even if full habitat protection is put into place, and

b) already in pristine condition have a significant probability of no change Scenario “0” assumes that current estimates of productivity parameters will remain constant in the future.

The *PATH Preliminary Decision Analysis Report* reported some preliminary comparisons of the effects of the “0” and the “B” scenarios. However, after this report was published, some concerns were raised about how the “B” scenario was implemented in the life-cycle model (*PATH Weight of Evidence Report*, Submission 4), and as a result all of the results presented in the *PATH Weight of Evidence Report* and in Section 2.2 of this report are only for the “0” habitat scenario.

We have addressed these concerns by presenting, in addition to results for both Habitat A and B scenarios, an alternative formulation of the “full protection” scenario, which we call Habitat C. The implementation of scenario C is analogous to the implementation in scenario 0, so these two scenarios are directly comparable. Similarly, habitat scenarios A and B are also directly comparable. As a technical note, the habitat improvement in scenario C is implemented by first calculating the expected change in the Ricker "a" parameter for each stock under Habitat A and again under Habitat B. Expected change is given by:

(%increase)*Pr(increase) - (%decrease)*Pr(decrease)

[17] As defined in Section 2.2, jeopardy probabilities are the probability that the number of spawners for a stock will exceed a defined survival or recovery threshold over a certain time period. The threshold is a certain number of spawners above which a stock is deemed to have survived and recovered under the ESA. There are two survival probabilities – one calculated over the first 24 years of the simulation period, the other calculated over the full 100-year simulation period. There are also two recovery probabilities – one calculated over 24 years, and the other over 48 years. Three of these probabilities — both survival and the 48-year recovery — are used as official standards by NMFS.

[18] These scenarios pertain only to freshwater spawning and rearing habitat. Sensitivity analyses to effects of habitat actions in the mainstem migration corridor and other potential habitat management options have been proposed for FY99.

The difference in the expected changes between Habitat B and Habitat A was used as the target expected change in Ricker "a" parameter for the Habitat C scenario, relative to the Habitat 0 scenario. The target was reached by altering Pr(increase) while keeping Pr(decrease) near a minimum. For example, if the expected improvement in Ricker a. from A to B was 0.2, this was implemented in scenario C, by adjusting the probabilities of increases and decreases so that the change in expected values was 0.2 relative to scenario 0. Table 2.3.1-1 shows the probabilities of increase and decrease in stock productivity parameters for each of the Habitat scenarios.

Table 2.3.1-1: Probabilities of future Ricker *a* values for seven Snake River spring/summer chinook stocks given three alternative scenarios of future habitat management. Prob(no change) means the probability that the *a* value does not change from its current (prospective simulation year 1) state by year 48 of the simulation. Prob(increase) and Prob(decrease) are interpreted similarly. The "increase" and "decrease" columns list the percent change in *a* value in the specified direction, relative to maximum likelihood estimates of the Ricker *a* values. Prob(12|increase) is the conditional probability that an increase occurs by simulation year 12, given that it occurs by year 48. The other conditional probabilities – Prob(*year*|*direction*) **-** have similar interpretations.

Stock	Prob (no change)	Relative Increase in a	Prob (increase)	Prob (12|increase)	Prob (24|increase)	Relative Decrease in a	Prob (decrease)	Prob (12|decrease)	Prob (24|decrease)
Option A									
Imnaha	0.9	12%	0.05	0.05	0.25	29%	0.05	0.25	0.5
Minam	1	0%	0	0	0	0%	0	0	0
Bear Valley	0.7	9%	0.15	0.1	0.4	28%	0.15	0.25	0.5
Marsh	0.85	11%	0.15	0.5	0.8	28%	0	0	0
Sulphur	1	0%	0	0	0	0%	0	0	0
Poverty Flats	0.5	14%	0.05	0	0.3	29%	0.45	0.35	0.5
Johnson	0.85	10%	0	0	0	28%	0.15	0.25	0.5
Option B									
Imnaha	0.85	12%	0.1	0.1	0.5	29%	0.05	0.25	0.5
Minam	1	0%	0	0	0	0%	0	0	0
Bear Valley	0.4	9%	0.6	0.2	0.8	28%	0	0	0
Marsh	0.85	11%	0.15	0.5	0.8	28%	0	0	0
Sulphur	1	0%	0	0	0	0%	0	0	0
Poverty Flats	0.6	14%	0.05	0	0.3	29%	0.35	0.35	0.5
Johnson	0.85	10%	0.07	0.4	0.8	28%	0.08	0.25	0.5
Option C									
Imnaha	0.88	12%	0.1	0.1	0.5	29%	0.02	0.25	0.5
Minam	1	0%	0	0	0	0%	0	0	0
Bear Valley	0.08	9%	0.92	0.2	0.8	0%	0	0	0
Marsh	1	0%	0	0	0	0%	0	0	0
Sulphur	1	0%	0	0	0	0%	0	0	0
Poverty Flats	0.79	14%	0.21	0	0.3	0%	0	0	0
Johnson	0.73	10%	0.27	0.4	0.8	0%	0	0	0

Figure 2.3.1-1 shows the effects of the habitat scenarios on average jeopardy probabilities for the 6th best stock. The comparisons shown below were run with a subset of combinations of other hypotheses to explore their effects. Equal weights were applied to the passage models, life cycle models, and extra mortality hypotheses to obtain the overall average jeopardy probabilities shown in Figure 2.3.1-1. The combinations used were:

Table 2.3.1-2: Combinations of hypotheses used for habitat sensitivity analyses.

Passage Model	Action	Passage Assumptions	Drawdown Assumptions	Life-cycle model	Extra Mortality
CRiSP, FLUSH	A1	Optimistic case	-	Alpha, Delta	BKD, Hydro, Regime Shift
CRiSP, FLUSH	A2	Optimistic case	-	Alpha, Delta	BKD, Hydro, Regime Shift
CRiSP, FLUSH	A3	Optimistic case	Optimistic case	Alpha, Delta	BKD, Hydro, Regime Shift

The relevant comparisons to make in Figure 2.3.1-1 are scenario 0 vs. scenario C, and scenario A vs. scenario B. Overall, the effects of the alternative habitat scenarios are relatively small. Effects are largest with the 48-year recovery probability. While the effects of habitat restoration (scenarios B and C) are positive for some actions and standards, there are a surprisingly large number of circumstances in which the effects of habitat restoration are negative. The explanation for this is that habitat improvements for a subset of the seven Snake River spring/summer stocks can interact with the in-river harvest schedules to produce complex patterns. When habitat improvements lead to larger escapements for stronger stocks, in some simulations this triggers higher in-river harvest rates for all Snake River stocks, including the weaker stocks (recall that in-river harvest rates increase as total Snake River abundance increases). As a consequence, in these simulations all stocks are harvested at a higher harvest rate which can lead to lower escapements for them than would otherwise be the case. This is particularly true for stocks with smaller abundances in pristine habitat that have low probabilities of increasing productivity with the habitat scenarios.

For example, the Bear Valley stock has the largest probability of an improvement in the Ricker "a" parameter for both scenarios B and C (Table 2.3.1-1). Bear Valley also has one of the larger spawning stocks historically (Beamesderfer et al. 1998). Therefore, increasing the productivity of the Bear Valley stock causes increased returns to the Columbia River mouth for the Bear Valley stock which can trigger a higher in-river harvest rate for all Snake River stocks. For a weaker stock like Sulphur that has a 0 probability of increasing under any of the habitat scenarios, this leads to higher harvest rates with no change in productivity, which reduces its projected spawning abundance.

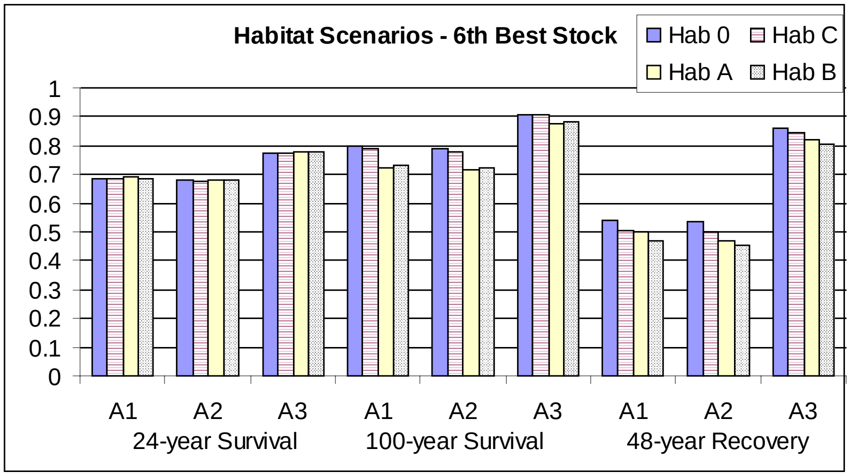


Figure 2.3.1-1: Effects of alternative habitat scenarios on average jeopardy probabilities for the 6th best stock.

The effects of habitat scenarios on individual stocks are shown in Table 2.3.1-3. Some stocks (e.g., Imnaha) show virtually no effect due to the habitat scenarios. The largest positive effects are for the Bear Valley stock, while the largest negative effects are for the Marsh Creek, Minam, and Sulphur stocks. Marsh Creek is the 6th best stock most often for the 24-year survival standard, while Minam is the 6th best stock most often for the 48-year recovery standard. Therefore, the results for these standards for the 6th best stock in Figure 2.3.1-reflect the negative effects shown for these stocks.

Table 2.3.1-3: Effects of alternative habitat scenarios on average jeopardy probabilities for Snake River spring/summer chinook index stocks (assuming a limited range of hypotheses).

Stock	Habitat Scenario	24-year Survival			100-year Survival			48-year Recovery		
		A1	A2	A3	A1	A2	A3	A1	A2	A3
Bear Valley	Hab 0	0.763	0.749	0.829	0.877	0.868	0.948	0.714	0.695	0.964
	Hab C	0.770	0.760	0.831	0.917	0.913	0.949	0.873	0.859	0.981
	Hab A	0.751	0.740	0.823	0.810	0.802	0.915	0.647	0.631	0.893
	Hab B	0.768	0.755	0.833	0.901	0.893	0.949	0.807	0.792	0.974
Imnaha	Hab 0	0.871	0.859	0.912	0.901	0.892	0.959	0.679	0.664	0.950
	Hab C	0.871	0.857	0.912	0.890	0.880	0.955	0.666	0.651	0.939
	Hab A	0.869	0.855	0.911	0.881	0.871	0.950	0.670	0.649	0.928
	Hab B	0.871	0.854	0.912	0.883	0.865	0.950	0.662	0.641	0.926
Johnson Flat	Hab 0	0.758	0.747	0.840	0.806	0.792	0.911	0.654	0.624	0.928
	Hab C	0.765	0.754	0.842	0.823	0.811	0.912	0.700	0.674	0.934
	Hab A	0.747	0.735	0.834	0.747	0.731	0.882	0.590	0.558	0.867
	Hab B	0.750	0.742	0.835	0.772	0.763	0.895	0.624	0.602	0.895
Marsh Creek	Hab 0	0.677	0.678	0.772	0.821	0.816	0.921	0.652	0.602	0.922
	Hab C	0.676	0.677	0.773	0.796	0.792	0.919	0.604	0.555	0.910
	Hab A	0.682	0.681	0.774	0.833	0.827	0.923	0.692	0.643	0.929
	Hab B	0.681	0.674	0.775	0.823	0.810	0.922	0.656	0.629	0.922

Stock	Habitat Scenario	24-year Survival			100-year Survival			48-year Recovery		
		A1	A2	A3	A1	A2	A3	A1	A2	A3
Minam	Hab 0	0.796	0.783	0.851	0.846	0.833	0.922	0.515	0.483	0.849
	Hab C	0.794	0.781	0.852	0.825	0.813	0.920	0.467	0.435	0.835
	Hab A	0.796	0.782	0.852	0.844	0.831	0.923	0.519	0.485	0.852
	Hab B	0.794	0.786	0.851	0.832	0.820	0.921	0.483	0.451	0.841
Poverty Flat	Hab 0	0.841	0.829	0.905	0.863	0.853	0.952	0.601	0.580	0.925
	Hab C	0.841	0.830	0.905	0.874	0.865	0.953	0.640	0.622	0.933
	Hab A	0.799	0.788	0.884	0.673	0.662	0.879	0.403	0.385	0.713
	Hab B	0.806	0.796	0.887	0.713	0.704	0.891	0.444	0.427	0.755
Sulphur Creek	Hab 0	0.641	0.626	0.724	0.735	0.729	0.843	0.529	0.539	0.824
	Hab C	0.639	0.624	0.723	0.713	0.707	0.839	0.483	0.496	0.812
	Hab A	0.640	0.625	0.722	0.733	0.728	0.843	0.529	0.535	0.828
	Hab B	0.636	0.627	0.722	0.716	0.706	0.841	0.487	0.477	0.822

Results in Figure 2.3.1-1 and Table 2.3.1-3 are averages over alternative passage models, life-cycle models, and extra mortality hypotheses. The effects of these individual hypotheses on responses to habitat scenarios are shown in Figure 2.3.1-2, using the A1 48-year recovery probabilities for Bear Valley for illustration. Overall, effects of habitat scenarios A, B,and C relative to scenario “0” are relatively constant among the extra mortality hypotheses, but differ somewhat between life-cycle and passage models. In particular, Hab B tends to result in larger differences in jeopardy probabilities relative to the Hab A scenario with the Delta and the FLUSH model than with the Alpha and CRiSP model.

In summary, alternative habitat scenarios lead to increases in average jeopardy probabilities for some stocks, and reductions for others. Overall effects were minor, and did not affect the ranking of actions. Because the analysis was done with a limited set of runs, definite conclusions about the affect of these habitat scenarios on the overall ability of actions to meet the standards (i.e., the fraction of runs in which all of the jeopardy standards are met) are not possible. However, for situations where average jeopardy probabilities are close to the standards with the base habitat scenario, such effects are possible. An analysis with a complete set of runs is required to fully address this question.

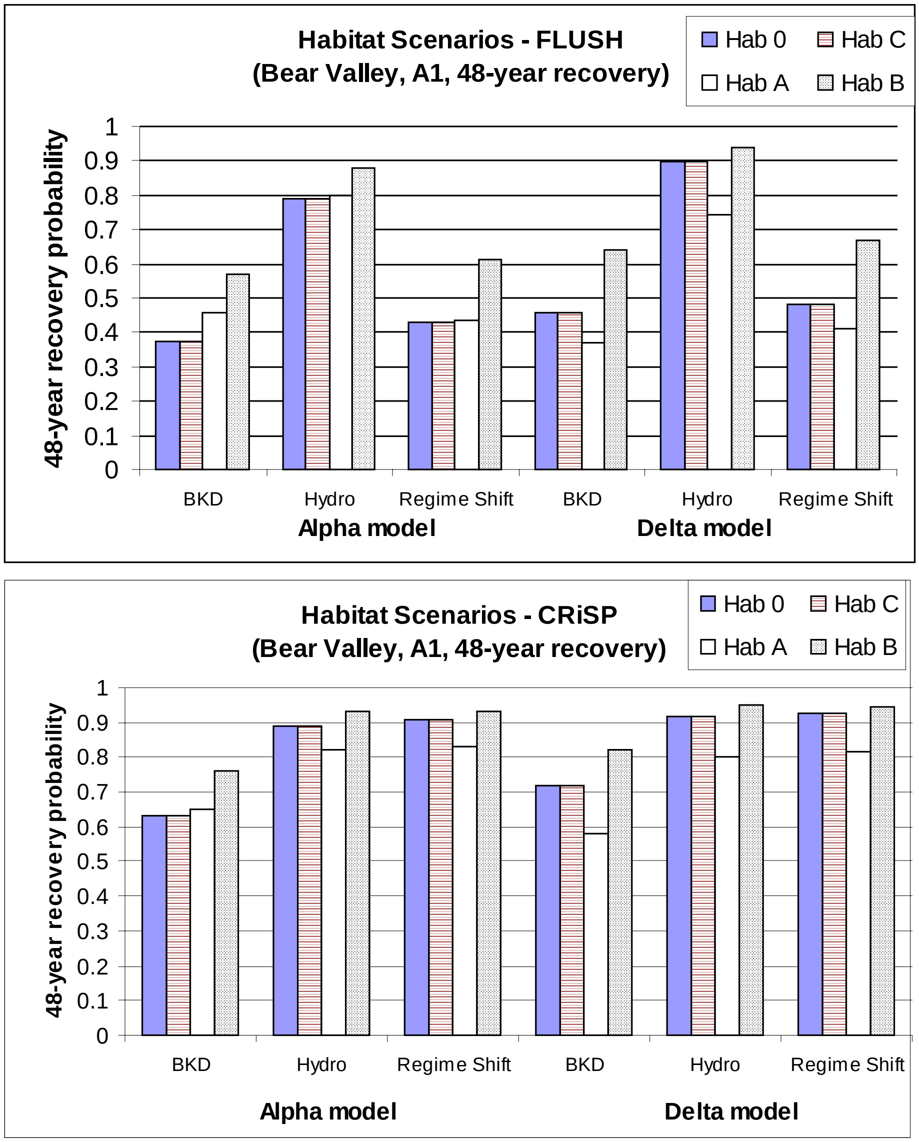


Figure 2.3.1-2: Effects of alternative habitat scenario on extra mortality hypotheses.

2.3.2 Effects of Alternative Harvest Scenarios

The current set of set of results incorporate mainstem and tributary harvest of spring and summer chinook. The total harvest rates for these fisheries applied in the model are based on an existing harvest management plan. In consultation with fishery managers, a PATH sub-group also defined two alternative harvest schedules that are based on the same stepped proportions of MSP as the current schedule:

a) conservative schedule, in which harvest rates for run sizes below the MSP level were divided by 1.5, and

b) a liberalized harvest schedule, in which harvest rates for run sizes below the MSP level were multiplied by 1.5.

The current and alternative harvest schedules are described in Section 4.3.7 of the *PATH Preliminary Decision Analysis Report*. In addition to the liberal and conservative harvest schedule described above, some PATH participants have also proposed a "low harvest" option, where the lowest harvest rate in the current schedule is applied at all run sizes. The mainstem harvest rate applied in the "low harvest" schedule is 3% for spring chinook stocks, and 2% for summer chinook stocks. The tributary rate is 5% for both groups of stocks at all run sizes. The low harvest option equates to the elimination of all mainstem commercial and sport fisheries and conservation level tribal ceremonial fisheries over the 100-year simulation period. The low harvest rate is hypothetical only and is merely intended as a further sensitivity analysis of outcomes to harvest schedules. To date, we have only completed analyses of all actions with the low harvest scenario using a limited set of runs. A sensitivity analysis of all actions using a full set of runs is in progress. These results will be included in the Update to the FY98 Final Report, which is scheduled for March 1999.

The current, liberal, conservative, and low harvest schedules are summarized in Table 2.3.2-1 and 2.3.2-2. The purpose of this section is to explore the effects of alternative harvest schedules on the overall results. For this sensitivity analysis, we again used a limited number of runs. The runs used were the same as those used for the habitat sensitivity analysis (Table 2.3.2-1), except that only the Delta model was used (the Alpha model was not set up to allow alternative harvest schedules).

Table 2.3.2-1: Spring chinook harvest schedules.

Run Size % of MSP	Current Harvest Management		Liberal Harvest Management		Conservative Harvest Management		"Low Harvest" Scenario	
	Mainstem	*Tributary*	*Mainstem*	*Tributary*	*Mainstem*	*Tributary*	*Mainstem*	*Tributary*
< 22%	0.055	0	0.083	0	0.037	0	0.03	0
22%-44%	0.082	0	0.123	0	0.055	0	0.03	0
45%-112%	0.14	0	0.210	0	0.093	0	0.03	0
113%-125%	0.25	0.05	0.25	0.05	0.25	0.05	0.03	0.05
126%-175%	0.3	0.15	0.3	0.15	0.3	0.15	0.03	0.05
176%-200%	0.35	0.2	0.35	0.2	0.35	0.2	0.03	0.05
>200%	0.4	0.25	0.4	0.25	0.4	0.25	0.03	0.05

Table 2.3.2-2: Summer chinook harvest schedules.

Run Size % of MSP	Existing Harvest Management		Liberal Harvest Management		Conservative Harvest Management		"Low Harvest Scenario"	
	Mainstem	*Tributary*	*Mainstem*	*Tributary*	*Mainstem*	*Tributary*	*Mainstem*	*Tributary*
< 25%	0.02	0	0.03	0	0.013	0	0.02	0
25%-49%	0.05	0	0.08	0	0.033	0	0.02	0
50%-99%	0.1	0	0.15	0	0.067	0	0.02	0
100%-129%	0.15	0	0.15	0	0.15	0	0.02	0
130%-149%	0.2	0.05	0.2	0.05	0.2	0.05	0.02	0.05
150%-169%	0.25	0.1	0.25	0.1	0.25	0.1	0.02	0.05
170%-200%	0.3	0.2	0.3	0.2	0.3	0.2	0.02	0.05
>200%	0.35	0.25	0.35	0.25	0.35	0.25	0.02	0.05

Effects of the different harvest schedules on the jeopardy probabilities are shown in Figure 2.3.2-1. The liberal and conservative harvest schedules have minor effects on the results relative to the current harvest schedule. The maximum increase in jeopardy probability due to reduction in harvest rates (i.e., the conservative harvest schedule) is 0.02, while the largest decrease in jeopardy probability with the liberal harvest schedule is 0.05.

The low harvest schedule leads to larger increases in jeopardy probabilities for A1 and A2 (maximum improvement for A1 or A2 is, but leads to decreases in jeopardy probability for A3, with the 100-year survival and the 48-year recovery standard. This seems counter-intuitive, but can be at least partially explained by the dynamics of the spawner-recruit relationship. This relationship has a dome-shape, so that the number of next generation progeny is reduced when there are large number of spawners (due to density-dependent interactions such as excessive crowding, competition, and cannibalism of juveniles). The current, liberal, and conservative harvest schedules are stepped so as to maintain quite high harvest rates at large run sizes. This reduces the number of spawners and helps to reduce the amount of density-dependent interactions. When harvest rates remain low at large run sizes, as they are in the low harvest scenario, excessive numbers of spawners can occur, which reduces the number of spawners produced in the next generation and, consequently, reduces the jeopardy probabilities. This can occur particularly for actions (like A3) and assumptions (like the Delta model) that produce larger number of spawners.

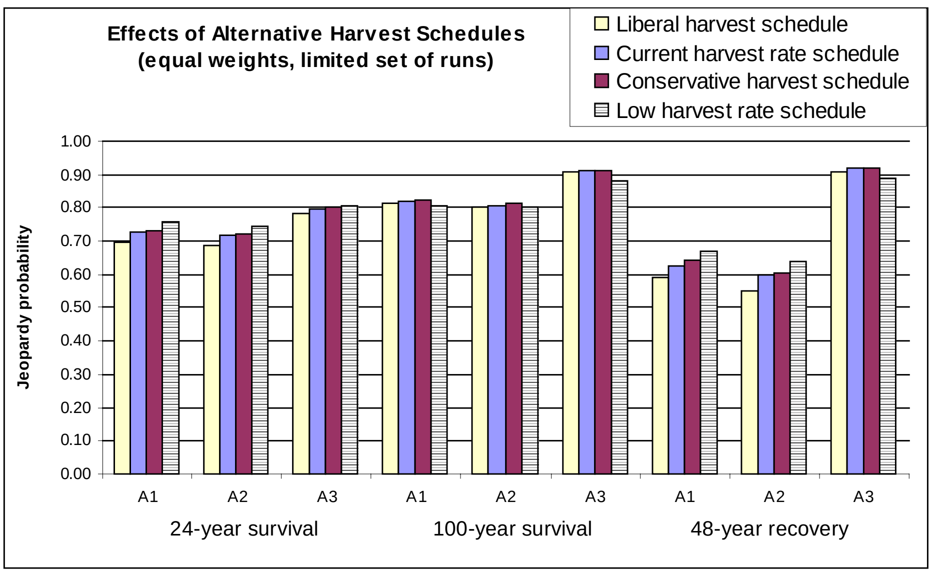


Figure 2.3.1-3: Effects of assuming alternative harvest schedule on jeopardy probabilities.

In summary, the effects of harvest rate reductions depend upon the size of the reduction and the specific step in the harvest rate schedule to which the reduction is applied. Small changes to harvest rates, such as those in the liberal and conservative harvest schedules have minimal effects on outcomes. Larger proportional changes in harvest rate have minimal effect at low harvest rate steps, but substantial reductions in harvest rates at higher steps (such as the low harvest scenario) can lead to improvements in projected escapements for actions that produce smaller projected numbers of spawners (such as A1 or A2), or reductions in projected escapements for actions or assumptions that tend to produce large numbers of spawners (such as A3 and the Delta model).

Changes in jeopardy probabilities were not sufficient to change the ranking of actions (A3 still produced higher jeopardy probabilities than A1 or A2). The limited set of runs used in this analysis does not allow general conclusions on whether alternative harvest scenarios affect the ability of actions to meet all of the standards. Although the effects of the reductions in harvest rates are small, they may be large enough to affect the ability of an action to meet the Jeopardy Standards. This is more likely to occur in situations where average jeopardy probabilities are already close to meeting the standards with the current harvest schedule (e.g., the weighted average 24-year survival probability is close to the standards of 0.7 for all actions; see Table 2.2.4-2). An analysis of all actions with a complete set of runs is needed to fully assess whether the harvest scenarios affect the ability of actions to meet all of the standards.

2.3.3 Effects of Additional Sources of Mortality

The intent in this analysis is to show the effects of explicitly incorporating sources of mortality that may not be reflected in the historical data. The current set of analyses uses historical spawner and recruit data up to brood year 1990 to estimate overall mortality. However, there are other sources of mortality that may not be reflected in this data. For example, predation on salmon smolts by Caspian terns and other bird predators in the estuary is hypothesized to have increased dramatically in the late 1980's and early 1990's. By projecting forward with data up to brood year 1990, the current set of runs implicitly assumes either that the mortality inflicted on salmon stocks by birds has not increased in recent times, or that bird mortality essentially replaced some other source of mortality that was operating through the historical period.

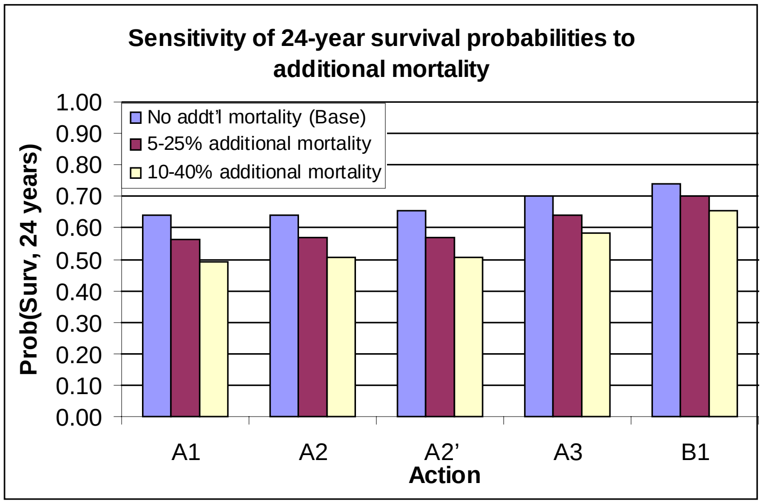
The effects of additional mortality sources were simulated by first defining a possible range of mortality rates to apply in the future, then selecting a mortality rate at random from within this range in each year of the forward projection, then applying this mortality rate to the projected recruits in that year. For example, if an additional mortality rate of 0.10 was selected, the projected number of recruits in that year was multiplied by 0.90. Note that this implementation assumes that the additional sources of mortality act on both transported and non-transported fish equally. This may or may not be the case; there are alternative hypotheses about whether barged fish are more or less vulnerable to bird predation in the estuary.

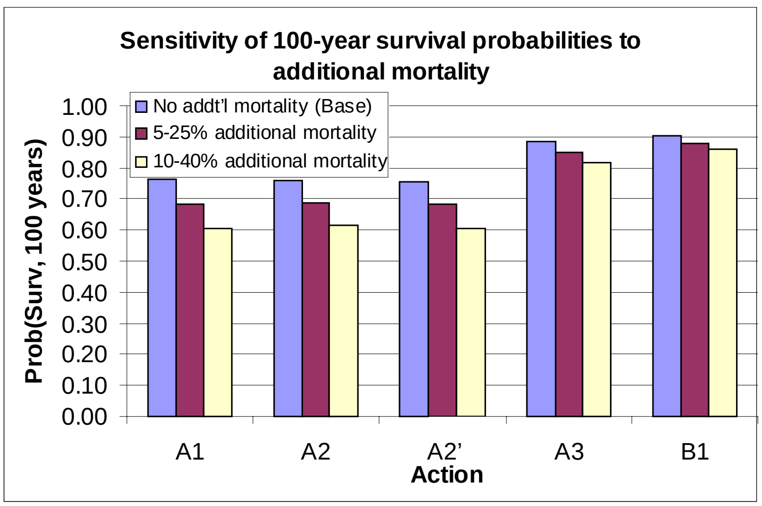
We looked at the sensitivity of the results to two ranges: 5 to 25%, and 10 to 40%. These ranges capture the low and high end of the range of estimates of the amount of mortality on spring/summer chinook smolts that is due to predation by birds in the estuary. These estimates range from an additional 5% mortality rate to 40%. The 5% estimate is derived from recoveries of PIT-tags on Rice Island (where there is a large concentration of birds). 5% of the tags from the release group were recovered here, which suggests that at least 5% of the release group were consumed by birds. This is a minimum estimate – there may have been more tags on the island that were not recovered because only a portion of the island was sampled for tags. The 40% estimate comes from recovery of radio-tags on Rice Island. This is a high estimate, because the added weight of the radio-tags may reduce the ability of smolts to avoid bird predators. An intermediate estimate of 25% is based on the results of a bioenergetics model, which estimates how many smolts birds need to eat. This estimate requires some assumptions about the proportion of smolts passing through the estuary that are wild spring/summer chinook.

Sensitivities to these ranges were explored using a single set of passage and (for A3 and B1) drawdown assumptions, selected so that the outcomes of these runs were close to the overall weighted average for that action (using equal weights on life-cycle model and extra mortality assumptions). For consistency, the results presented below also apply equal weights to the life-cycle and extra mortality hypotheses.

Decreases in jeopardy probabilities are largest with the 48-year recovery probability (Figure 2.3.3-1). Overall, the maximum decrease in any jeopardy probability for a 5-25% range of additional mortality was 0.15, and for a 10-40% range was 0.23 (both maximum decreases were for action A1, 48-year recovery probability). The smallest decrease was 0.02 with a 5-25% range of additional mortality and 0.05 with a 10-40% range (both for action B1, 100-year survival probability).

The simulated effects of an additional mortality affect all actions relatively equally, and thus do not affect the ranking of actions. Insofar as this limited set of runs is representative of the average of all runs (recall that the runs were selected so as to approximate the weighted average jeopardy probabilities over all runs), additional mortality sources do affect the ability of all actions to meet the 24-year survival standard, and the ability of A1, A2, and A2' to meet the 100-year survival standard. However, an assessment of all of the runs is needed to draw these conclusions with confidence.





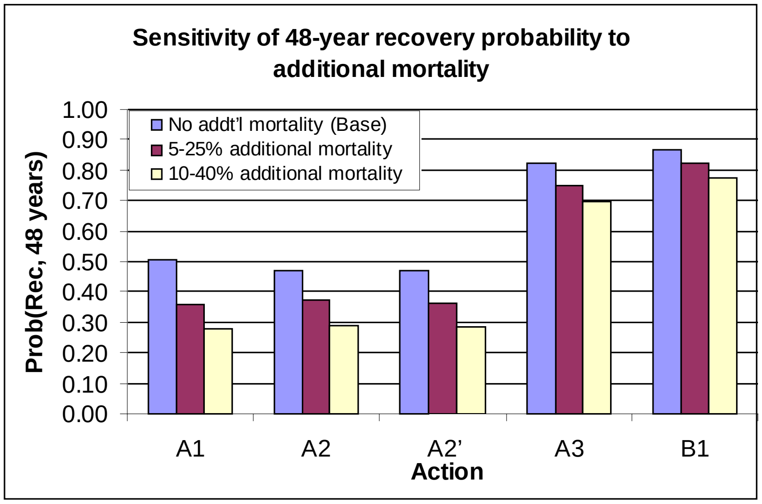


Figure 2.3.3-1: Sensitivity of average jeopardy probabilities to two ranges of additional mortality.

To see if the general patterns shown above are influenced by the particular passage model, life-cycle model, or extra mortality hypothesis, results are shown separately for these hypotheses in Table 2.3.3-1. Only results for the 48-year recovery probability are shown to illustrate general patterns; these patterns are similar with the other survival and recovery probabilities. The results suggest that patterns are approximately constant over hypotheses, although relative declines tend to be somewhat greater in FLUSH than in CRiSP, and somewhat greater in the BKD extra mortality hypothesis than in Hydro or regime shift.

Table 2.3.3-1: Effects of additional mortality on 48-year recovery probabilities.

Hypothesis	Additional Mortality Assumption	48-year Recovery Probability				
		A1	A2	A2'	A3	B1
CRiSP	Base	0.627	0.628	0.615	0.785	0.820
	5-25%	0.496	0.529	0.519	0.725	0.752
	10-40%	0.415	0.446	0.435	0.669	0.696
FLUSH	Base	0.383	0.317	0.327	0.864	0.920
	5-25%	0.222	0.217	0.207	0.771	0.891
	10-40%	0.145	0.130	0.132	0.720	0.853
Alpha	Base	0.502	0.442	0.462	0.821	0.856
	5-25%	0.323	0.354	0.352	0.735	0.812
	10-40%	0.255	0.270	0.279	0.695	0.775
Delta	Base	0.508	0.504	0.481	0.827	0.883
	5-25%	0.395	0.392	0.374	0.761	0.831
	10-40%	0.306	0.305	0.289	0.695	0.774
BKD Extra mortality	Base	0.379	0.355	0.365	0.743	0.800
	5-25%	0.231	0.233	0.251	0.639	0.728
	10-40%	0.155	0.155	0.169	0.571	0.663
Hydro Extra mortality	Base	0.596	0.573	0.530	0.889	0.928
	5-25%	0.450	0.486	0.431	0.828	0.897
	10-40%	0.367	0.385	0.353	0.787	0.863
Regime Shift Extra mortality	Base	0.575	0.507	0.566	0.858	0.893
	5-25%	0.434	0.429	0.450	0.806	0.857
	10-40%	0.357	0.359	0.376	0.757	0.821

2.3.4 Increase in Adult Conversion Rates Following John Day Drawdown

In addition to the effects of John Day + Snake River drawdown on juvenile passage, PATH explored the effects of B1 on adult upstream passage. Two alternatives for these effects were modeled. The first alternative was to assume that there would be no increase in adult conversions through the John Day reach. This alternative uses the same conversion rates as A3, so it still considers the effects of Snake River dam drawdown on adult passage through the Snake River reach. The second alternative was to assume that the conversion rate through the John Day reach following drawdown would increase to 1.0 (1986 to 1995 mean per-project conversion rate is 0.937 for spring chinook, 0.918 for summer chinook).

We compared the two conversion rate alternatives for a subset of B1 runs that included best-case and worst-case combinations of hypotheses about B1 timing, (i.e., best = short pre-removal and transition period; worst = long pre-removal and transition period), and all possible equilibrium juvenile survival rates, passage models, extra mortality hypotheses, and life-cycle models. Average, minimum, and maximum differences between the two alternatives are shown in Table 2.3.4-1. Differences were calculated as jeopardy probabilities with boosted conversion rates — jeopardy probabilities with non-boosted conversion rates. These average differences are close to zero, indicating that conversion rates have minimal influence on results. Negative differences sometimes arise because small positive or negative fluctuations in outcomes can result from random processes within the life-cycle model. Therefore, very small positive effects of increased conversion rates will sometimes be masked by negative random fluctuations.

These small effects are not likely to change the overall ranking of actions. An analysis with a full set of runs would be needed to thoroughly assess how these effects would change the fraction of B1 runs meeting all of the standards.

Table 2.3.4-1: Average, minimum and maximum differences in jeopardy probabilities arising from assuming an increase in conversion rate through the John Day reach following drawdown of John Day dam.

		24-year Survival	100-year Survival	24-year Recovery	48-year Recovery
FLUSH	**Ave.**	0.00	0.00	0.02	0.01
	Min	-0.03	-0.01	-0.02	-0.03
	Max	0.02	0.02	0.05	0.10
CRiSP	**Ave.**	0.00	0.00	0.01	0.03
	Min	-0.01	-0.01	-0.03	-0.01
	Max	0.03	0.02	0.06	0.11

3.0 Fall Chinook

PATH devoted a considerable amount of time to fall chinook analyses in FY98. Results of these analyses are documented in this section of the FY98 Final Report. Major accomplishments (and their associated section in this report) are:

- A hydro workgroup compiled and reviewed the available data on hydrosystem effects on juvenile fall chinook. (Section 3.1.1)
- Run reconstructions were completed for four fall chinook stocks, providing spawner-recruit data for life-cycle modeling (Section 3.1.2)
- Fall chinook passage models were updated with the data sets compiled by the hydro workgroup (Section 3.2.1)
- A fall chinook version of the PATH life-cycle model (fall BSM) was developed (Section 3.2.2)
- The passage models and life-cycle models were used to generate retrospective estimates of survival measures (Sections 3.3.1 and 3.3.2)
- A set of assumptions for prospective analyses was developed, and used to project outcomes of management actions A2, A2', A3, and B1 (Section 3.4)

3.1 Data

3.1.1 Passage Data/Estimates for Model Calibration, Validation, and Analysis

The evaluation of the effects of the hydrosystem on juvenile wild Snake River fall chinook is in large part determined from the passage models. These passage models estimate the relative impact of several factors such as predation and direct dam mortality on juvenile chinook populations during the migration starting from the head of LGR pool through to the BON tailrace. The PATH Fall Chinook Data Workgroup has reviewed a number of data sources that could potentially be used to develop the passage models. Some of these data sources were incorporated into the passage models, while others were not. The purpose of this section of the report is to document the data sources that were actually used in the passage models. Other information not presented may be useful in describing alternatives to current model configurations.

Passage Data

The CRiSP and FLUSH modeling groups incorporated much of the most up-to-date pertinent information regarding passage and migration behavior of juvenile fall chinook in the Snake River. The data and estimates that were used by the modelers are presented in the "Passage Data" section. The information in this section is partially a distillation of technical memoranda that were authored by members of the Fall Chinook Hydro/Passage Modeling Work Group. Those documents are archived on a WEB page maintained by University of Washington staff.

Physical Data

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

Flow, Spill, and Reservoir Elevation Data

Both fall chinook passage models require two sets of daily flow, spill, and elevation files, one for the retrospective simulations and one for the prospective simulations. The retrospective simulation are based on historic flow, spill, and elevation data and the prospective simulations are based on output from hydroregulation models that describe how flows and spills would vary from the historic under different flow management scenarios such as A1 (the 1995 Biological Opinion), A2 (maximize transportation) and A3 (drawdown to natural river).

Historic flow, spill, and elevation data are archived by the Army Corps. Monthly flow data from 1929 through 1989 are available for all projects, or, prior to their construction, estimates of flows at project sites. These are referred to as “regulated” flows because they were controlled (regulated) through operations at storage reservoirs (BPA 1993a). These flows and elevations are used within the passage models to generate water travel times.

Both regulated and unregulated flows are “monthly” except that April and August are split into two halves because these are frequently transition months in which changing conditions would render monthly means less useful for power planing purposes. Thus the data are presented in 14 rather than 12 periods.

Hydroregulation model output used in the prospective simulations is in the same format. These models manipulate the operations of storage reservoirs in the U.S. and Canada to try and meet flow, spill, and elevation targets proposed in various management options. The output of interest to modelers is the flow, spill, and elevations at all eight Columbia and Snake River dams for each of the 14 periods from either 1929 through 1989 or from 1929 through 1978 depending on the hydroregulation model.

Because the passage models operate on a daily basis, daily flow, spill, and elevation data are required. These data have been compiled in electronic formats and are maintained by various agencies including the Fish Passage Center. The data are collected daily although some days or even months are missing, mostly in the winter months. Data for both flow, spill, and elevations at all eight Columbia and Snake River hydro projects is available from the 1965 through the present. Daily data may be used directly for retrospective simulations. For prospective simulations the daily flow and spill data provide templates so that the “monthly” (14 period) data can be modulated to reflect the variability in day to day operations. Modulation is described for each passage model later in this report.

Temperature Data

Daily, dam specific temperature data are available from 1965 through the present. These data also are in electronic format and are maintained by various agencies including the Fish Passage Center. Temperature data can affect predation rates, initial emigration dates, and fish travel times in the passage models.

Initial Emigration Timing

Both passage models operate on a daily time step. In the model, the initial emigration distribution represents the relative size a daily cohort of fish and the date their migration begins at the head of LGR pool. This initial relative distribution was constructed from the CPUE by date of seined wild or wild and hatchery fish used in the PIT-tagging studies conducted by the USFWS between 1991-1997 (Connor et al. 1993, 1994a, 1994b, 1996, 1997, 1998). The development of the initial distributions in both models is defined in the model descriptions section of this chapter.

Fish Travel Time Estimates of Snake River Fall Chinook

Investigators at NMFS and USFW have collected data describing the fish travel times (FTT) of Snake River fall chinook. The data are based on PIT-tagged individual fish; information is archived in PTAGIS. Both wild and hatchery fish have been tagged and released upstream from LGR Dam. There are a variety of ways these data can be grouped temporally and spatially (reach length) to yield response indices of interest. Later in this report, passage modelers will describe how they selected, partitioned, and treated these data.

NMFS used Lyons Ferry Hatchery stock to estimate survival and migration rates for juvenile fall chinook throughout the hydro-system during the years 1995-1998 (Smith et al. 1997 and Muir et al. 1998). PIT-tagged fish were released at several sites upstream from LGR Dam. Experimental groups were liberated from late May through mid-June. Fish Travel Times (FTT) could be estimated between any release site and LGR by the difference between release date and detection date. Subsequent detections were possible at other dams, since PIT-tagged fish were diverted from the collection systems and returned to the tailrace to continue their migration seaward. In 1995 and 1996, McNary was the terminal detection site. In 1997 and 1998, PIT detectors installed at Bonneville Dam extended the observational window through the entire system.

Since 1991, the USFWS has been PIT-tagging naturally produced juvenile fall chinook in the Snake River upstream from LGR Dam. Fish were collected with beach seines tagged and liberated at the same locations as the hatchery stocks. During the years 1991-1994 the objective was to characterize the early life history of wild pre-migrant in the reach upstream from LGR Dam. In 1995-1998, the effort also provided a means to compare the survival and migration rate of these wild fish to the hatchery counterparts used by NMFS (Connor et al. 1997, Connor et al. 1998) (Table 3.1.1-1).

Wild fish were also detected at MCN and later at BON projects. Estimation of FTT for wild fish became increasingly difficult at detector sites further downstream. This occurred as the number of surviving fish decreased as they traveled downstream and differences between collector efficiencies at the different projects. For example, of the 123 wild PIT-tagged fish detected at LGR in 1997only 10 were detected at MCN, and none were detected at BON.

Table 3.1.1-1: Dam sites where migration rates were estimated by investigators.

Year	Wild	Hatchery	Citations
1991	LGR	--	Connor et al. 1993
1992	LGR	--	Connor et al. 1994a
1993	LGR	--	Connor et al. 1994b
1994	LGR	--	Connor et al. 1996
1995	MCN	MCN	Connor et al. 1997, Smith et al. 1997
1996	MCN	MCN	Connor et al. 1998, Muir et al. 1998
1997	BON	BON	Muir and Smith Preliminary 1998 results – WEB page

As many of the wild and hatchery experimental fish have been tagged and released as parr, still in their rearing phase, it is difficult to interpret the relevance of migration rates based on elapsed time observed from the release site to arrival at LGR Dam. Thus, the time it takes fish to move to LGR Dam reflects both rearing and migratory phases and associated behaviors. Expectedly, the observed FTT are highly variable and can be sensitive to size at release, date of release, and environmental conditions such as water temperature and river discharge. The information does not permit clear partitioning of migratory behaviors associated with the rearing and active migrant phases. To date, both modeling groups have modeled the migration and the migration plus rearing phase using different techniques to distinguish these different phases (see Section 3.2.1). Once a fish passes LGR reservoir they are assumed to be in the migration phase.

Reservoir Survival

Loss of sub-yearling chinook to predators is the primary source of mortality in the reservoirs modeled in the passage models (CRiSP also included a small amount of nitrogen mortality – see Section 3.2.1 for a description of how this was modeled in CRiSP). The fish community, and the species of predators in particular, has changed considerably in the last 100 years. Li et al. (1987) and Poe et al. (1994) discussed how introductions have greatly changed the predator community in the Snake and Columbia rivers. Prior to predator introductions (before 1900), northern pikeminnow (formally called northern squawfish), white sturgeon, bull trout, cutthroat trout, and sculpins were likely the major predators in the system. After introductions and hydroelectric development, the list of major predators is northern pikeminnow, walleye, smallmouth bass, channel catfish, and sculpins (Poe et al. 1991, Vigg et al. 1991, Rieman et al. 1991). The exotic species, bass, walleye, and channel catfish, have undoubtedly increased over the last 100 years, primarily since impoundment (Li et al. 1987), while white sturgeon, bull trout, and cutthroat trout are less abundant.

Predator abundance and consumption rates have been suggested to increase after the installation of dams (Poe et al 1991, Poe et al. 1994). These changes are thought to have occurred because: slow water habitat preferred by these predators has increased (Poe et al. 1994); dam induced stress, injury and disorientation have increased smolt vulnerability (Ledgerwood et al. 1990, 1994); and increases in temperature have increased the energetic demands of these predators (Poe et al. 1991, Vigg et al. 1991). The most pronounced influence of dam operations on the effects of predation is observed in the boat-restricted zone (BRZ) of dam tailraces. Densities and consumption rates of pikeminnows are much higher than that observed elsewhere in the reservoirs (Vigg et al. 1991, Rieman et al. 1991, Petersen et al. 1990).

Data on predator abundance and consumption rates is extensive for JDA Reservoir between 1982-1986 (see Poe and Reiman 1988). A monitoring program has estimated the abundance and consumption for pikeminnow, walleye, smallmouth bass, and catfish relative to JDA estimates since 1991 (Zimmerman and Parker 1995, Ward 1997). The data available for parameterizing predator abundance and predator consumption rates in the passage model is limited to a portion of the time series analyzed. Therefore, the passage models had to assume that predator dynamics have not changed over the time-series analyzed.

Predator Abundance

Within John Day reservoir Beamesderfer and Rieman (1991) estimated the population abundance of key predatory fish species using a multiple recapture model (Table 3.1.1-2). Predator abundance estimates are highly variable, however, these estimates are thought to be conservatively low (Beamesderfer and Rieman 1991).

Table 3.1.1-2: Population abundance estimates of key predatory fish species using a multiple recapture model as reported by Beamesderfer and Rieman (1991). 95% confidence limits in parentheses.

Year	N. Squawfish(>250mm)	S. Bass(>200mm)	Walleye(>250mm)
1984	69,947 (55,250-86,040)	- -	13,043 (6,573-23,006)
1985	84,114 (66,905-105,749)	31,948 (18,967-44,929)	18,426 (7,236-39,855)
1986	102,888 (75,215-136,059)	37,959 (29,019-46,899)	14,036 (4,520-36,003)
Average	85,316 (65,693-106,645)	34,954 (25,166-44,741)	15,168 (6,067-32,914)

Gut analysis of catfish suggests that they feed heavily on juvenile salmonids (Vigg et al. 1991), however, mark-recapture estimates were not performed for this species. This analysis was not possible partly due to the low numbers observed in JDA. While catfish may not be present in large numbers in the Columbia, gillnet CPUE suggest that catfish densities in the Snake River relative to other species are high (Zimmerman and Parker 1996).

Mark-recapture predator abundance estimates have not been conducted for other years or in other reservoirs. A predator monitoring program, however, has estimated the relative abundance (the abundance index-AI) based on CPUE of predators in several reservoir since 1990 (Ward et al. 1998). Passage models used this information to describe the relative abundance of predators in the Snake and Columbia Rivers. The AI has been determined for the BRZ, mid-reservoirs, and forebays for several reservoirs and allows site specific information in the passage models. This information is important as predation rates vary greatly between these sections with greatest disparity observed between the BRZ and the mid-reservoir (Petersen 1994).

A sensitivity on the impact of the predator removal program is currently addressed in PATH. The predator removal program began in 1990. Relative predator abundance used in the model included AIs from 1990 and 1991, the years during the monitoring program thought to be the least impacted by the predator removal program and most similar to the intensive JDA predator studies.

Predator Consumption Rates

Another important component needed to assess the impact of predators on prey is the predator consumption rates for specific prey items. Diets of these major predators have switched to include a greater proportion of juvenile salmonid in slow water reservoirs and in dam tailraces (Buchanan et al. 1981, Brown and Moyle 1981, Poe et al. 1994, Tabor et al. 1993). Temperature also has a large effect on predator consumption rate (Kitchell et al. 1977). The increase in temperature due to impoundment likely further increases the consumption rate on juvenile salmonids.

During the predator study between 1982-1986 conducted in JDA, gut analysis were performed on the four major predators previously discussed (Vigg et al. 1991). Consumption rates were estimated using a method derived from Swenson and Smith (1973). These consumption estimates allow further investigation on the effects of prey density (functional responses), temperature, and location on predation rates. Bioenergetics models that describe physiological limits have also been developed for these predators (Hewett and Johnson 1992, Vigg et al. 1991, Petersen and Ward, in press). This information is crucial in the development of mechanistic models used to describe the impact these predators have on juvenile fall chinook.

Currently, the USGS Biological Research Division (BRD) is conducting studies to determine the influence of shoreline structure, temperatures, and water velocities on predator dynamics. These studies are being conducted in free-flowing sections in the Snake and Columbia Rivers as well as in reservoir habitat. The impact that dams have on habitat alteration is also being investigated through historic channel mapping. These studies will elucidate how habitat changes from the hydroelectric system alter predator impacts on juvenile salmonids.

Direct Dam Survival

Juvenile salmonids pass a dam through one of three routes of passage; through turbines, spill, or bypass and sluiceway systems. Mortality associated with each of these routes of passage has been determined in various studies and are applied to the passage models to account for direct dam mortality. The relative proportion of a daily cohort of fish apportioned to each of these routes is dependent on spill rates, spill effectiveness (SS), and fish guidance efficiencies (FGE).

<u>Turbine Survival</u>

The proportion of smolts entering the turbines is based on the proportion of the flow not spilled and the proportion of smolt not diverted into the bypass systems (1-FGE). We define turbine survival as the proportion of fish surviving direct turbine passage injuries. Turbines also have an indirect effect on fish survival by causing stress, injury, and disorientation thereby increasing vulnerability to predation. However, this is accounted for in the passage models by applying a BRZ specific predation rate. Turbine survival studies published through 1990 at Snake and lower Columbia River dams have been reviewed by Iwamoto and Williams (1993). Turbine survival estimated in the nine studies ranged from 80-98% and averaged 90%. Estimates of direct mortality of subyearling chinook through turbine units include: 3.9% at Bonneville PH1 (Weber 1954); 11% mortality at McNary Dam (Schoeneman et al. 1961); and 13% mortality at John Day Dam (Raymond and Sims 1980). Gilbreath et al. (1993) 3-year average turbine mortality of subyearling chinook at Bonneville PH2 of 2.3% direct plus 6.8% indirect mortality near the outfall.

The passage models use a turbine survival estimate of 0.90, which was the same estimate applied to spring/summer chinook in PATH analyses. The above information suggests variability exist in the estimates (2.3-13% mortality). The WG also recommend sensitivities to $\pm$ 0.03 (0.87 and 0.93) to reflect the general variability in empirical estimates.

<u>Spill Survival</u>

The ISG (1996) and Whitney et al. (1997) reviewed estimates of spill survival in the Snake and Columbia Rivers published through 1995. Nearly all of these studies involved steelhead or yearling chinook salmon. Mortality estimates for 10 of the 13 studies ranged between 0-0.022. Estimates for the other three studies were extremely variable (0.04 to 0.275) and, in our opinion, should be viewed with caution. In some studies, mortality appears to be higher in spillbays with spill deflectors than in those without deflectors, but these differences are generally not statistically significant (e.g., Muir et al. 1995). Additional studies by the Corps of Engineers are currently underway to resolve this issue. Recent studies conducted at The Dalles Dam involving paired releases of subyearling fall chinook above the spillways and below the tailrace indicate that the spill survival of summer-migrating may be affected by changes in spill (Dawley and Gilbreath 1998, AFEP presentation). The workgroup has currently agreed on a value of 0.98 as the mortality experience through the spillway. There are other studies that show different values from this, but the workgroup has not yet reached agreement on how applicable these studies are.

Spillway Passage Efficiency and Effectiveness

This topic was first surveyed for PATH for use in spring/summer chinook passage modeling in Appendix 4 of Chapter 6 (Marmorek et al. 1996). Since that paper was drafted there have been additional reports published that estimate smolt passage at spillways throughout the Columbia Basin.

We identify two measures to describe spill passage. Johnson et al. (1997a) defines spill efficiency (SY) as the proportion of the smolt population passing the entire dam that migrates through the spillway. Spill effectiveness (SS) is the ratio of SY to the proportion of total flow that is discharged as spill.

The PATH spring/summer yearling chinook passage model analyses used a spill effectiveness of 1.0 for most dams in the Snake and lower Columbia Rivers. This was based on a review of estimates available from investigations published through 1995. Typically, those early investigations depicted considerable variability associated with the spillway passage estimates, which made it difficult to statistically demonstrate departure from the conventionally assumed SS value of 1.0. However, a number of those studies suggested SS may often exceed 1.0.

Recently increased research effort has been conducted at some dams in the Snake (Table 3.1.1-3) and Columbia River (Tables 3.1.1-4 and 3.1.1-5), and those results are available for consideration and incorporation into passage model analyses/sensitivities. The most prominent studies are hydroacoustic based investigations that incorporate a new more quantitatively rigorous estimation procedure.

Table 3.1.1-3: Spill passage estimates acquired at Snake River dams since 1995.

Dam	Citation	Year	Tool	Spp.	SY(%)	SS	Spill(%)
LGR	Johnson et al. (1997b)	1997	HA	mix	48	1.45	33
	Johnson et al. (1997c)	1997	TE	ST	41		
LGO	Muir et al. (1998)	1997	PIT	ST	41	1.24	
LMO	Johnson et al. (1997a)	1997	HA	mix	69	1.9	
IH	Eppard et al.(1997)	1997	TE	FC	82	1.2	>65

Lower Granite Dam

Spring migrants: At Lower Granite Dam hydroacoustic sampling occurred only during the spring migration, thus estimates for summer migrating subyearling fall chinook are not available. Overall, during the study period 48% of the smolts were estimated to pass the dam via spill. Spill averaged about 33% of total discharge during that time, yielding an SS estimate of 1.45 (Johnson et al. 1997b). This estimate pertains not to a particular species but the entire spring-migrating smolt population that was dominated by hatchery steelhead during that year. These estimates are consistent with telemetry-based estimates for yearling chinook reported by Wilson et al. (1991); with SS = 1.5 at 40% spill.

Telemetry-based estimates of SY are consistent with those obtained using hydroacoustics. Johnson et al. (1997c) reported that 41% of the radio-tagged hatchery steelhead passed through the spillway during the spring study period that roughly coincided with the hydroacoustic evaluation. Spill effectiveness estimates were not reported for the telemetry data.

Little Goose Dam

Spring Migrants: The only estimate describing spill passage at this site is reported in a recent draft report released by NOAA Fisheries (Muir et al. 1998). For hatchery steelhead that were used in a survival study, they estimated that 41% passed the dam via spill (SY), yielding a spill effectiveness (SS) estimate of 1.24 at the prevailing spill proportions.

Lower Monumental Dam
Spring and Summer Migrants: Johnson et al. (1997a) used hydroacoustic sampling to estimate SY and SS at Lower Monumental Dam in 1997. This information offers the rare opportunity to compare estimates for spring and summer-migrating smolts. Conditions at Lower Monumental Dam in 1997 were such that a considerable volume of spill was provided during an 11-day sampling period in mid-June, at a time when subyearling chinook dominated the fish population passing the dam. This provided a unique opportunity to acquire estimates for this chinook race with hydroacoustic sampling.

Results indicate that over the entire sampling period (April 21-June 2, and June 12-25) 69% of the smolts passed through the spillway. Overall, spill effectiveness over the entire sampling period was estimated at 1.9.

Summer migrants: The investigators did not calculate separate estimates for the spring and June sampling periods. However, they did note that based on visual inspection of graphs, SY appeared higher during June when subyearling chinook dominated the run, and SS was similar to that observed for spring migrants, at near 1.9.

Ice Harbor Dam
Fall Chinook: In 1997, hatchery-reared subyearling fall chinook were tagged with radio transmitters to identify their passage routes at Ice Harbor Dam (Eppard et al. 1997). Investigators estimated that 82% of the smolts passed the dam via spill, yielding a spill effectiveness estimate (SS) of 1.2 at the prevailing high spill levels (>65%).

Lower Columbia River Dams
Giorgi and Stevenson (1995) surveyed smolt passage route estimates obtained at John Day, The Dalles and Bonneville Dam up through 1994. Over the previous years, few spillway passage investigations had been conducted at TDA and JDA and none at Bonneville Dam or McNary Dam.

From 1986-1989 summer hydroacoustic monitoring was conducted at John Day Dam (Reviewed by Giorgi and Stevenson 1995). Seasonal SS estimates were reported and ranged from 1.0 to 1.4. However, it is not clear to what extent the targets truly represented fall chinook in the pelagic multi-species fish population prevalent during the summer.

Summer Migrants: At The Dalles Dam, Giorgi and Stevenson (1995) noted that a hydroacoustic study (Steig and Johnson 1986) produced a graph that indicated an SS of approximately 2.0 over a range of spill from about 10-20%, for summer migrants.

Since 1994, subsequent to the Giorgi and Stevenson (1995) review, a number of telemetry-based smolt passage investigations have been staged at lower Columbia River projects that permit assessments of spillway usage. Formal estimates of spill effectiveness are not typically calculated. But authors do report season-wide spill efficiency (SY) estimates and sometimes report the percent river flow discharged as spill. Inspection of those values in the following Tables 3.1.1-4 and 3.1.1-5 indicates that spill effectiveness regularly exceeds 1.0 at John Day, The Dalles and Bonneville dams.

Table 3.1.1-4: Spring Migration: Lower Columbia projects- Telemetry-based estimates of spill efficiency. Where spill % is not indicted, estimates were not provided in the reports that were available for our inspection at the time of this review.

Dam	Citation	Year	Spp	SY (%)	Spill (%)
JD	Sheer et al. (97)	1995	YC	28.7	4-5
	Holmsberg et al. (97)	1996	YC	42	17-32
	Hensleigh et al. (97a)	1997	YC	64-66	
			ST	51-53	
TD	Sheer et al. (97)	1995	YC	75	50-60
	Holmsberg et al. (97)	1996	YC	76	
	Hensleigh et al. (97a)	1997	YC	74	
			ST	78	
BON	Holmsberg et al. (97)	1996	YC	63	30-65
	Hensleigh et al. (97a)	1997	YC	75	
			ST	72	

Table 3.1.1-5: Summer Migration: Lower Columbia projects- Telemetry-based estimates of spill efficiency. Where spill % is not indicted, estimates were not provided in the reports that were available for our inspection at the time of this review.

Dam	Citation	Year	Spp	SY (%)	Spill (%)
JD	Holmsberg et al. (97)	1996	FC	40	12-20
	Hensleigh et al. (97a)	1997	FC	45-50	
TD	Sheer et al. (97)	1995	FC	74	63-67
	Holmsberg et al. (97)	1996	FC	66	
	Hensleigh et al. (97a)	1997	FC	74	
BON	Holmsberg et al. (97)	1996	FC	40	30-57
	Hensleigh et al. (97a)	1997	FC	26	

Prescribing SS in Fall Chinook Passage Model Analyses

The Hydro Passage Work Group recognized that the emerging information indicates that SS regularly exceeds 1.0 for all species, including summer-migrating fall chinook at dams where it has been evaluated. However, there is not yet sufficient information to describe an SS X Spill (%) relationship at all dams. As a consequence, for initial fall chinook passage model analyses we adopted 1.0 (the same value used previously in the spring chinook analyses) as the default value for SS at all dams except The Dalles. We suggest that a factor of 2.0 be applied at The Dalles Dam at spill levels ≤ 30% and suggest that above 30% spill, the relationship grades from 2.0 to 1.0 according to Equation (1). This relationship predicts a factor of 1.5 at 65% spill.

$$P_f = 2.0 * P_w \qquad 0 < P_w \leq 0.30 \qquad \text{[Eqn. 1]}$$
$$P_f = (2.43 - 1.43 * P_w) * P_w \qquad P_w > 0.30$$

P_f is the proportion of fish passing over the spillway and P_w is the proportion of total river flow passing over the spillway. Spill efficiency, as we have defined it, is (P_f □P_w). Support for this relationship comes from Giorgi and Stevenson (1995) who cited three investigations that estimated spill efficiency from which SS estimates could be derived.

The WG recommended conducting a sensitivity analysis, following the initial passage model analyses. In that sensitivity SS at all four Snake River Dams would be defined by the same function. The suggested sensitivity analysis for Snake River projects relies on a relationship for spring chinook salmon at Lower Granite Dam that is based on radio-telemetry observations (Wilson et al. 1991). Using the radio-telemetry estimates of SS and forcing the relationship through zero, and asymptotic at 100% spill, the following relationship (2) from Smith et al. (1993) can be applied:

$$P_f = 2.583*P_w - 3.250*P_w^{\,2} + 1.667*P_w^{\,3} \qquad \text{[Eqn. 2]}$$

where P_f is the proportion of fish passing over the spillway, and P_w is the proportion of water passing over the spillway.

<u>Bypass and Sluiceway Survival</u>

The mortality of fish that pass a dam via the bypass systems was determined by experiments conducted by NMFS during 1995-1997 (Muir et al. 1998). These tests consisted of paired releases of subyearling fish into the bypass system at Little Goose Dam and into the river immediately below the tailrace. These experiments were conducted in 1995, 1996 and 1997 but the experiments in 1995 and 1996 were deemed less reliable due to temperature and handling problems than in 1997 . Therefore, the 1997 value only (0.88; S. Smith, pers. comm.) was used for both bypass and sluiceway survival in the current set of passage model analyses.

Because of the nature of the research (i.e., paired releases), the survival rate reflects both the direct mortality that occurs as fish pass through the dam, but also the mortality associated with bypass related predation in the tailrace. Attempts to model subyearling survival must acknowledge that tailrace mortality is included in the estimate and devise a method of partitioning.

<u>Fish Guidance Efficiency</u>

The proportion of juvenile salmonids entering the bypass is determined by the fish guidance efficiency (FGE) of screens used to divert the juveniles away from turbines. Two sets of FGEs were modeled developed for fall chinook to provide an opportunity to examine model sensitivity to two assumptions about the effectiveness of Extended Length Bar Screens (ELBSs). The first sensitivity assumed that the FGEs remained at the same level reported for Submersible Traveling Screens (STSs) while the second set of FGEs assumed an increase in FGEs for the ELBSs. The two sets are presented in Tables 3.1.1-6 and 3.1.1-7. The data are presented and discussed in Krasnow (1997).

Table 3.1.1-6: Year and Project specific FGEs for sensitivity number one (no increase in FGEs with Extended Length Bar Screens).

Year/ Dam	LGR	LGS	LOMO	ICE	MCN	JDA	TDA	BONN
1965				0	0		0.02	0
1966				0	0		0.02	0
1967				0.03	0		0.02	0
1968				0.03	0	0.02	0.02	0
1969			0.02	0.03	0	0.02	0.02	0
1970		0.02	0.02	0.03	0	0.02	0.02	0
1971		0.12	0.02	0.03	0	0.02	0.02	0.4
1972		0.12	0.02	0.03	0	0.02	0.02	0.4
1973		0.37	0.02	0.03	0	0.02	0.02	0.4
1974		0.37	0.02	0.03	0	0.02	0.02	0.4
1975	0.09	0.37	0.02	0.03	0	0.02	0.46	0.4
1976	0.09	0.37	0.02	0.03	0	0.02	0.46	0.4
1977	0.27	0.24	0.02	0.03	0	0.02	0.46	0.4
1978	0.27	0.37	0.02	0.03	0	0.02	0.46	0.4
1979	0.27	0.37	0.02	0.03	0.05	0.02	0.46	0.4
1980	0.27	0.37	0.02	0.2	0.03	0.02	0.46	0.4
1981	0.27	0.37	0.02	0.2	0.03	0.02	0.46	0.4
1982	0.27	0.37	0.02	0.2	0.22	0.02	0.46	0.4
1983	0.27	0.37	0.02	0.4	0.24	0.02	0.46	0.38
1984	0.27	0.37	0.02	0.4	0.24	0.02	0.46	0.46
1985	0.27	0.37	0.02	0.4	0.24	0.34	0.46	0.46
1986	0.27	0.37	0.02	0.4	0.24	0.34	0.46	0.46
1987	0.27	0.37	0.02	0.4	0.24	0.34	0.46	0.46
1988	0.27	0.48	0.02	0.4	0.24	0.34	0.46	0.46
1989	0.27	0.48	0.02	0.4	0.24	0.34	0.46	0.45
1990	0.27	0.48	0.02	0.4	0.24	0.34	0.46	0.45
1991	0.49	0.48	0.02	0.4	0.24	0.34	0.46	0.45
1992	0.49	0.48	0.49	0.4	0.24	0.34	0.46	0.45
1993	0.49	0.47	0.49	0.46	0.24	0.34	0.46	0.47
1994	0.49	0.47	0.49	0.46	0.24	0.34	0.46	0.47
1995	0.49	0.48	0.49	0.46	0.24	0.34	0.46	0.47
1996	0.49	0.48	0.49	0.46	0.24	0.34	0.46	0.47
1997	0.49	0.45	0.49	0.46	0.37	0.34	0.46	0.47
1998	0.49	0.45	0.49	0.46	0.37	0.34	0.46	0.47

Table 3.1.1-7: Year and Project specific FGEs for sensitivity number two (FGEs increase with Extended Length Bar Screens).

YEAR/ DAM	LGR	LGS	LOMO	ICE	MCN	JDA	TDA	BONN
1965				0	0		0.02	0
1966				0	0		0.02	0
1967				0.03	0		0.02	0
1968				0.03	0	0.02	0.02	0
1969			0.02	0.03	0	0.02	0.02	0
1970		0.02	0.02	0.03	0	0.02	0.02	0
1971		0.12	0.02	0.03	0	0.02	0.02	0.4
1972		0.12	0.02	0.03	0	0.02	0.02	0.4
1973		0.37	0.02	0.03	0	0.02	0.02	0.4
1974		0.37	0.02	0.03	0	0.02	0.02	0.4
1975	0.09	0.37	0.02	0.03	0	0.02	0.46	0.4
1976	0.09	0.37	0.02	0.03	0	0.02	0.46	0.4
1977	0.27	0.24	0.02	0.03	0	0.02	0.46	0.4
1978	0.27	0.37	0.02	0.03	0	0.02	0.46	0.4
1979	0.27	0.37	0.02	0.03	0.05	0.02	0.46	0.4
1980	0.27	0.37	0.02	0.2	0.03	0.02	0.46	0.4
1981	0.27	0.37	0.02	0.2	0.03	0.02	0.46	0.4
1982	0.27	0.37	0.02	0.2	0.22	0.02	0.46	0.4
1983	0.27	0.37	0.02	0.4	0.24	0.02	0.46	0.38
1984	0.27	0.37	0.02	0.4	0.24	0.02	0.46	0.46
1985	0.27	0.37	0.02	0.4	0.24	0.34	0.46	0.46
1986	0.27	0.37	0.02	0.4	0.24	0.34	0.46	0.46
1987	0.27	0.37	0.02	0.4	0.24	0.34	0.46	0.46
1988	0.27	0.48	0.02	0.4	0.24	0.34	0.46	0.46
1989	0.27	0.48	0.02	0.4	0.24	0.34	0.46	0.45
1990	0.27	0.48	0.02	0.4	0.24	0.34	0.46	0.45
1991	0.49	0.48	0.02	0.4	0.24	0.34	0.46	0.45
1992	0.49	0.48	0.49	0.4	0.24	0.34	0.46	0.45
1993	0.49	0.47	0.49	0.46	0.27	0.34	0.46	0.47
1994	0.49	0.47	0.49	0.46	0.26	0.34	0.46	0.47
1995	0.5	0.48	0.49	0.46	0.24	0.34	0.46	0.47
1996	0.53	0.48	0.49	0.46	0.34	0.34	0.46	0.47
1997	0.53	0.45	0.49	0.46	0.68	0.34	0.46	0.47
1998	0.53	0.45	0.49	0.46	0.68	0.34	0.46	0.47

Transportation

A portion of the subyearling chinook collected in the bypass collection facility at LGR, LGO, LMO, and MCN are transported. The proportion of fish entering the collection facility is determined by the FGE. The transport start and stop dates and the probability of being transported during the collection period determine the proportion collected that is transported. This information was reported prior to 1982 by NMFS and subsequently from Army Corps (Table 3.1.1-8). The proportion of the fish collected that were transported may not represent the proportion of the migratory population transported as a large portion of the migratory population may arrive at a collector project after the stop date. Thus, the total proportion of the migratory population transported is determined in the passage models and is dependent not only on the probability of being collected and transported at a specific project but also on the arrival date at that project.

Table 3.1.1-8: Cutoff dates for transporting fall chinook smolts at Lower Granite (LGR), Little Goose (LGO), Lower Monumental (LMO), and McNary Dam (MCN).

Year	LGR	LGS	MCN
77	6/13	6/15	
78	6/19	6/13	8/30
79	7/2	6/18	8/22
80	7/5	7/2	9/3
81	7/28	7/23	9/9
82	7/27	7/20	9/22
83	7/28	7/6	9/20
84	7/24	7/26	9/26
85	7/21	7/21	9/24
86	7/22	7/1	9/24
87	7/29	7/7	10/27
88	7/29	7/13	9/19
89	7/25	7/9	9/17
90	7/24	7/19	9/12
91	10/29	10/29	10/29
92	10/29	10/30	12/5
93	10/30	10/30	10/28
94	10/30	10/30	11/20
95	10/30	10/30	12/10
96	10/29	10/26	12/13
97	11/8	11/2	12/12

Fish that are transported either through trucks or barges incur some mortality before release below BON. Studies designed to estimate transport survival on subyearling chinook have not been conducted, and hence a value of 0.98 was adopted from the yearling chinook passage model. However, studies estimating yearling chinook transportation survival have also not been conducted. The value of 0.98 was suggested by PATH representatives from the Army Corp of Engineers and is based on anecdotal evidence and visual observations. The WG recommended conducting a sensitivity analysis on this value, following the initial passage model analyses.

No transport studies have been conducted estimating post-release survival of subyearling Snake River chinook. However, a series of evaluations have been conducted at MCN Dam using a mixed wild/hatchery population arriving at that dam (Giorgi 1998). Those transportation experiments were conducted at MCN Dam beginning in 1978 continuing through 1983, then again in 1986-1988. As the stock evaluated in these studies consist mainly of wild and hatchery populations emanating from the mid-Columbia (Hanford) their applicability to Snake River subyearling fall chinook is uncertain. We propose to look at this further in FY99. Therefore, the post-BON survival of transported fish is presently analyzed in the life-cycle models (Section 3.2.2).

Reach and Project Survival Estimates

The above topics discuss data used to develop passage model components. These models were used to estimate and predict changes in survival throughout the juvenile migration from the head of LGR pool to tailrace of BON. Studies investigating survival of subyearling chinook can thus be used to calibrate or validate these models.

NMFS has been estimating the survival of juvenile fall chinook through portions of the lower Snake River since 1995. Those investigations used Lyons Ferry Hatchery fish. PIT-tagged fish were released at a number of sites upstream from LGR Dam. They estimated survival through a mark-recapture method from each release site to the tailrace of LGR Dam and through subsequent projects to the tailrace of LMO Dam. Capture and survival probabilities were estimated with the software program SURPH using the single release model. Detailed results are presented in Smith et al. (1996), Muir et al. (1997), and Muir and Smith (1998).

Survival – Release to LGR

Each year, over a seven-week release period the survival decreased steadily through time. Survival decreased considerably from near 70% in late May to near 5% for releases made in early July (Figure 3.1.1-1). Data were detailed in tables presented in Muir and Smith (1998).

Survival – LGR to LOMO

Survival estimates were calculated for weekly blocks of fish passing LGR. Survival to LOMO was similar in 1995 and 1996, in terms of magnitude and pattern. In 1997 survival estimates were substantially lower than previous years, particularly in July and early August (Figure 3.1.1-2). Data and estimates are presented in Muir and Smith (1998).

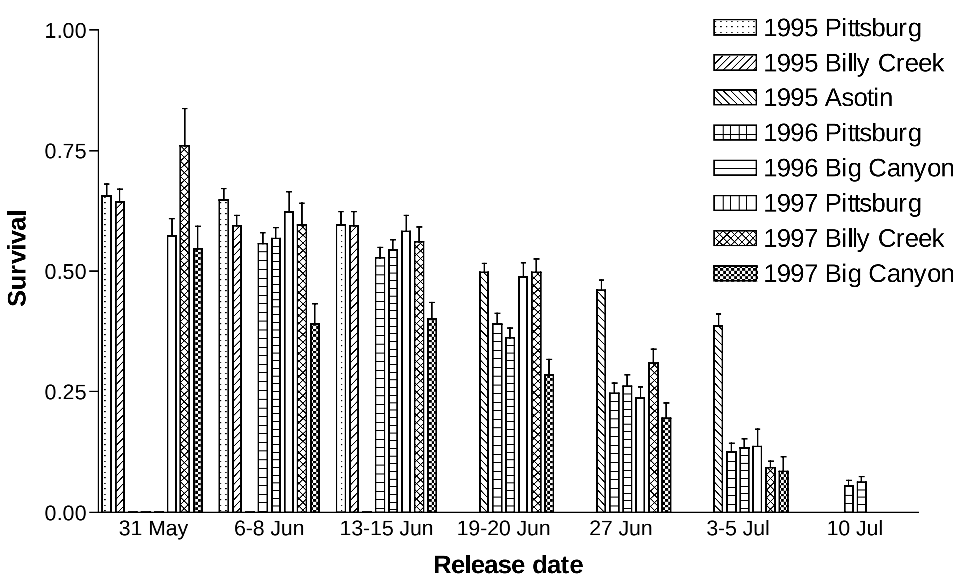


Figure 3.1.1-3: Survival from point of release in the Snake (Pittsburg Landing, Billy Creek, and Asotin) and Clearwater (Big Canyon Creek Rivers to the tailrace of Lower Granite Dam for 1995, 1996, and 1997 releases. Standard errors are also shown.

Figure 3.1.1-1 reproduced from Muir and Smith (1998) draft manuscript submitted to PATH that is archived on the fall chinook WEB page.

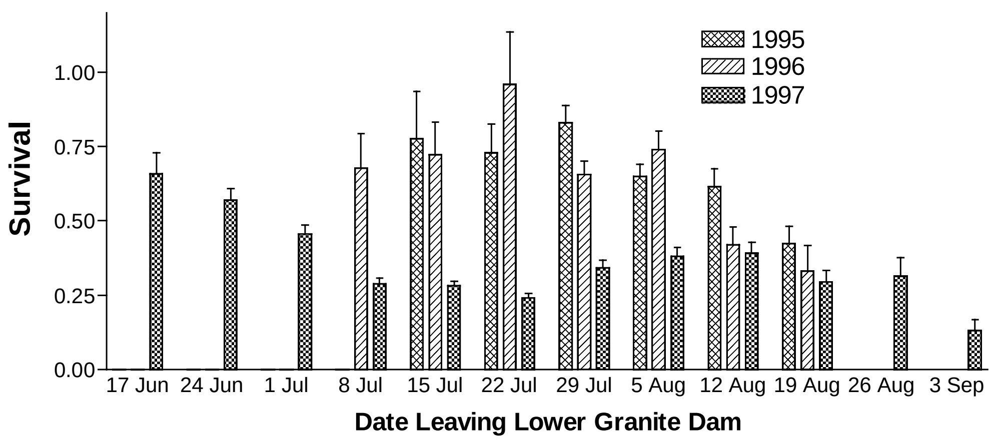


Figure 3.1.1-2: Survival estimates for PIT-tagged hatchery subyearling fall chinook salmon leaving Lower Granite Dam to the tailrace of Lower Monumental Dam each week during 1995-1997. Standard errors are also shown.

Figure 3.1.1-2 reproduced from Muir and Smith (1998) draft manuscript submitted to PATH that is archived on the fall chinook WEB page.

Summary

1. Daily site-specific river flows, spills, reservoir elevation, and temperature are physical data incorporated into the passage models. This information is archived in electronic format by various organizations including the Fish Passage Center. Hydroregulation models developed by the Army Corps of Engineers estimate flows, spills, and elevations for project sites under different management options.

2. The initial emigration distribution defines the size of a daily cohort and when they begin their migration. These distributions were determined from seining catch rates for sub-yearling chinook collected for PIT-tag studies. These distributions vary from year to year and may be affected by environmental variables. Whether a subyearling is in a rearing/migratory or migratory phase may also influence these distributions.

3. PIT-tag studies provide estimates of wild and hatchery Snake River fall chinook fish travel time available for constructing FTT relationships used in the passage models. Those estimates extend to MCN dam from 1995 to 1997 and to BON Dam in 1997. Estimates through LGR Pool are available for 1991-1997.

4. Predation is the primary source of mortality in the reservoirs described in the passage models. Predation rates are determined from predator abundance and consumption estimates. These variables have been estimated through an intensive predation study in JDA from 1982-1986 and from a predator-monitoring program conducted in the Lower Snake and Columbia Rivers. Predation estimates derived from these studies are applied to the entire time-series analyzed in the passage models.

5. Direct turbine mortality is assumed to be 10% based on the average survival estimates from several studies. These studies exhibit variability in survival, perhaps warranting a sensitivity on this estimate.

6. Based on a review of several studies, the Work Group suggested that a spillway survival value of 98% be used for the current round of passage model analyses. The workgroup needs to assess the applicability of other studies that produce different estimates of spill survival.

7. Several studies indicate that spill effectiveness is near 1.0, and this value had been adopted in the passage models. Recent estimates acquired over the last three years indicate that spill effectiveness at most sites exceeds 1.0. Future analyses need to provide a contrasting assessment using the most current information.

8. Based on a 1997 study at Little Goose Dam, the workgroup adopted a bypass survival of 0.88 for the current round of passage modeling. Further analyses of other estimates and sensitivity analyses to a range of values are needed.

9. FGE, transport start and stop dates, and probability of being transported after collection determine the proportion of fish transported at collector projects. NMFS and the Army Corps of Engineers have compiled this information since 1975. Transported fish have an assumed survival of 98% upon release below BON. No studies, however, have been conducted to determine this survival estimates suggesting a sensitivity to different survival assumptions is necessary. No post-release transport evaluations have been conducted using fall chinook at Lower Granite Dam and therefore are currently estimated in the life-cycle model.

10. Survival estimates obtained during the years 1995-1997 for Snake River fall chinook juvenile salmon (Lyons hatchery stock) in the Lower Snake, are possibly the only data available for use in either the calibration or validation of the passage model. Appropriate survival values for this process, to-date, have not been agreed upon by the workgroup. The workgroup is discussing how to use these estimates for calibration or validation.

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3.1.2 Run Reconstruction Information

Introduction

The objective of this report is to present spawner and recruit data, and a brief evaluation of temporal and spatial patterns of stock productivity and survival for Snake River fall chinook and other naturally spawning stocks with similar life history characteristics. The results of this report are for use in the PATH (Plan for Analyzing and Testing Hypotheses) retrospective and prospective analyses of alternative hydro-management actions in the Columbia and Snake rivers. The four stocks we performed run reconstructions for are: 1) Snake River brights (SRB); 2) Hanford Reach-Yakima River upriver brights (HYURB); 3) Deschutes River brights (DES); and 4) North Fork Lewis River brights (NFL) (Figure 3.1.2-1). These four stocks all share common characteristics of out-migrating as sub-yearling (ocean type life history), spawning some time after returning to natural areas (brights), spawning in expansive mainstem or larger second order tributary areas, and using spawning and rearing habitat in areas with flows regulated by upstream dams. We did not include the tule fall chinook populations that return to tributaries in the lower Columbia River (below The Dalles Dam) at a younger age and are ready to spawn upon return. Tules are exposed to different harvest impacts compared to brights (due to different spatial and temporal return patterns) and the vast majority of tule chinook are from hatchery origin.

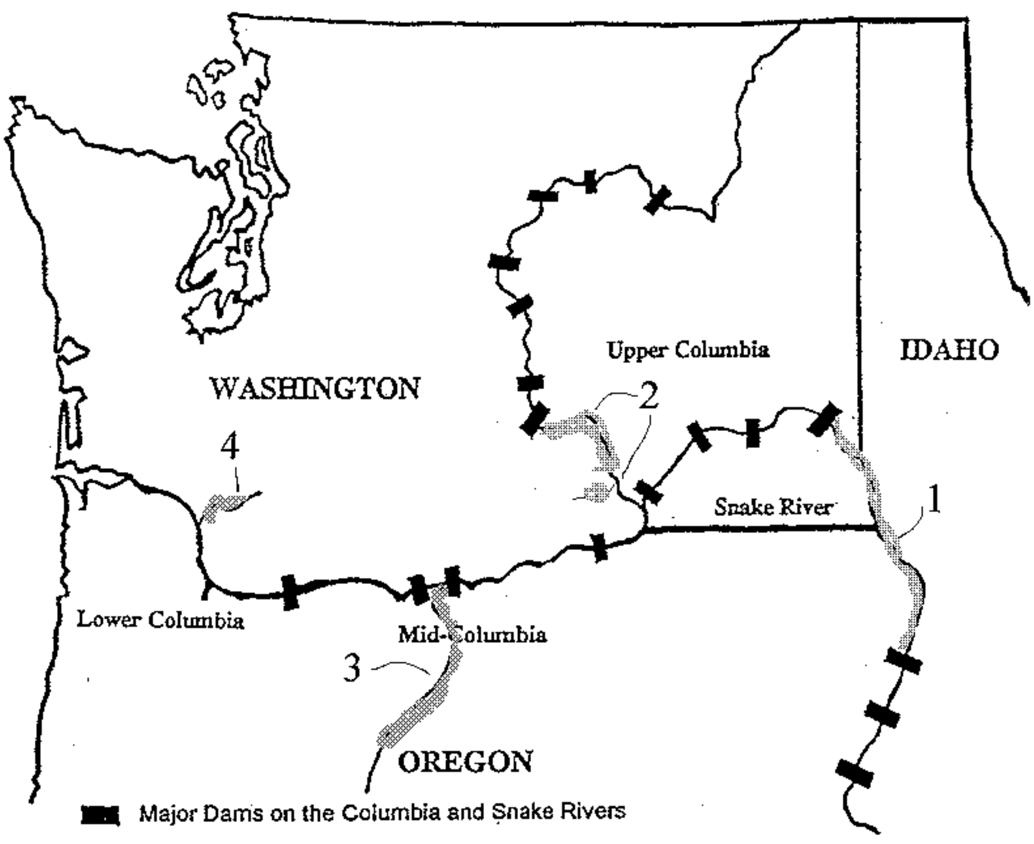

Figure 3.1.2-1: Location of bright fall chinook salmon index populations: 1. SRB; 2. HYURB; 3. DES; 4. NFL.

The SRB population consists of all adult fall chinook presently spawning downstream from the Hells Canyon Dam complex to the uppermost dam on the lower Snake River (Figure 3.1.2-1). The current distribution of SRB chinook is confined to the mainstem below Hells Canyon dam. In the era before Snake River hydro development, the fall chinook spawned in the mainstem Snake River from the mouth to Shoshone Falls, a distance of 984 river kilometers (Rkm). Historically, the majority of SRB fall chinook spawning occurred above the present Brownlee dam site (Haas 1965; Howell et. al. 1985). The existing naturally spawning fall chinook population is a remnant of a formerly large run that returned an average of 41,000 spawners annually from 1957 to 1960. The SRB fall chinook migrates a minimum of 720 Rkm past eight mainstem dams of the Snake and Columbia rivers (Figure 3.1.2-2). The mainstem reach presently accessible to spawning adults is 232 Rkm in length (Table 3.1.2-1). Quality of habitat for SRB spawners and juveniles is considered poor to fair relative to habitat used by the other three index stocks (Table 3.1.2-1). Hatchery influences have been highly variable ranging from no hatchery influence (brood years 1964-82) to proportionally large numbers of hatchery fish escaping to spawning areas (BY 1983-91) (Figure 3.1.2-3). For the 1988 return year, all returning fish of natural origin were trapped and used for hatchery brood stock and the stock origin of naturally spawning fish was 100% hatchery fish. Presently, all coded-wire-tagged (CWT) fish are removed at Lower Granite Dam and the number of hatchery fish that spawn naturally is very low. During the 1964-91 brood years, the harvest of fall chinook in the Snake River has been virtually non-existent. The fall chinook population was listed as threatened under the Endangered Species Act (ESA) in May 1992; therefore, Columbia River mainstem harvest management is currently guided by ESA requirements for SRBs. The brood year age 4 in-river (freshwater) harvest rates, and cumulative ocean exploitation rates, have fluctuated around 30% over the period of this analysis (Figure 3.1.2-4).

Table 3.1.2-1: Index populations of wild bright fall chinook salmon in the Columbia and Snake River basins.

Subbasin, *Management designation*	**Years of Complete Data**	**Current # Dams Passed**	**Ocean Distance (km)**	**Available Habitat (km)**	**Habitat Quality**	**Hatchery Influence on Adult Spawners (pre-1992) Level and Source (average)**
Snake River above LGR *Snake River Bright (SRB)*	1964-96	8	720	232	Poor-fair	Highly variable 3% local hatchery
Columbia River above PR *Upriver Bright (URB)*	1964-96	4	540	79	Good	Variable 93% local hatchery
Deschutes River *Upriver Bright (URB)*	1977-96	2	342	167	Fair-good	Extremely low 0% local hatchery
North Fork Lewis River *Lower River Wild (LRW)*	1964-96	0	160	13	Fair-good	Low 4% local hatchery

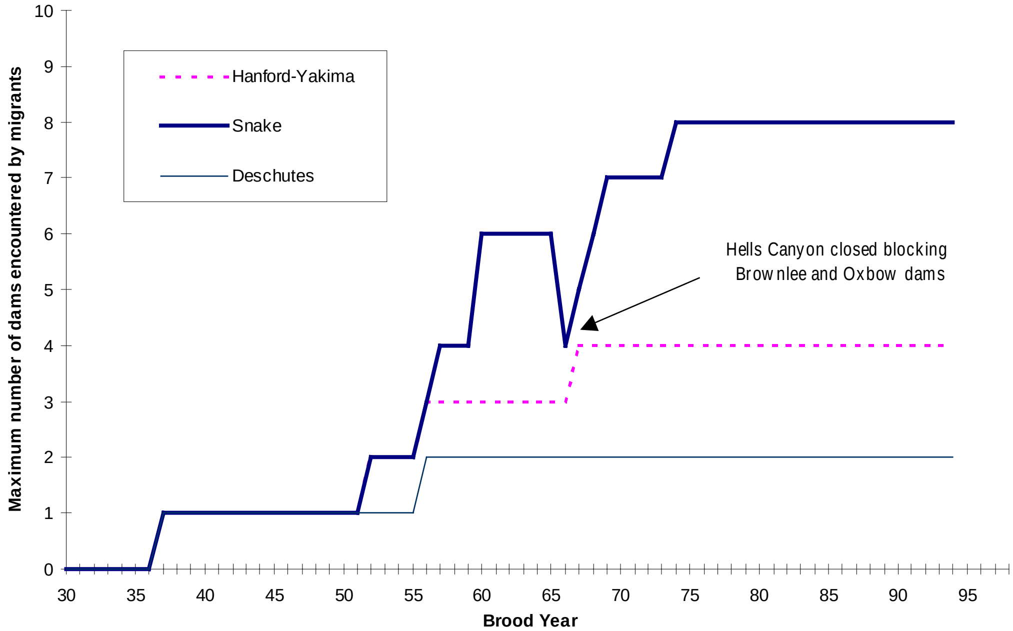

Figure 3.1.2-2: Maximum hydropower dams encountered by fall chinook sub-yearling out-migrants (and returning adults) from the three stocks above Bonneville Dam.

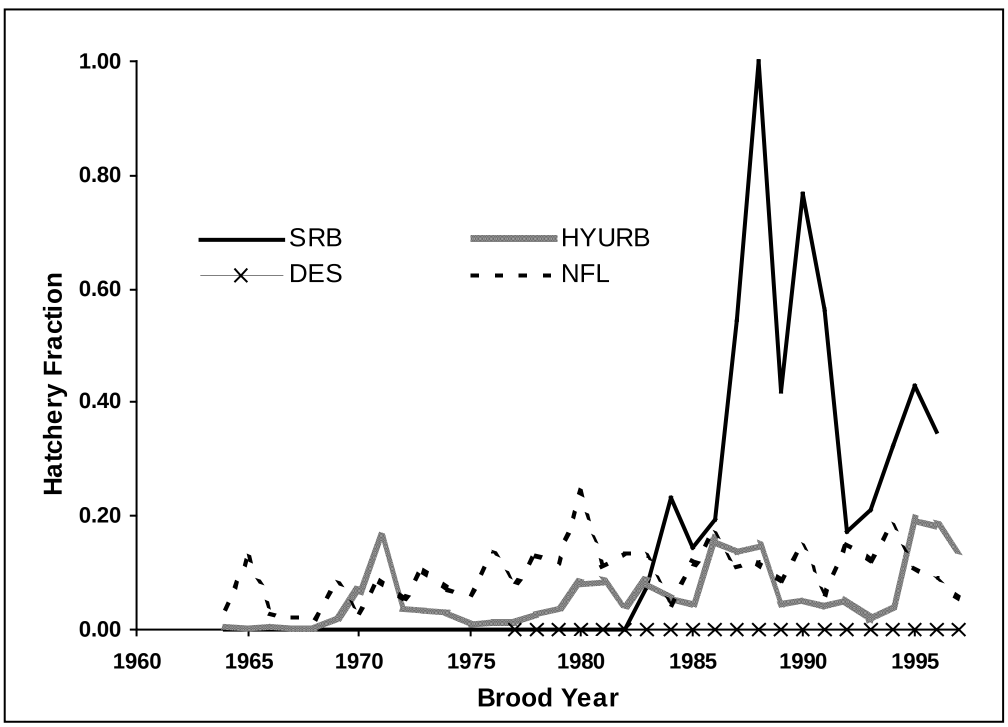

Figure 3.1.2-3: Proportion of natural adult and jack spawners that are composed of hatchery strays, as opposed to progeny of natural spawners, 1964-97.

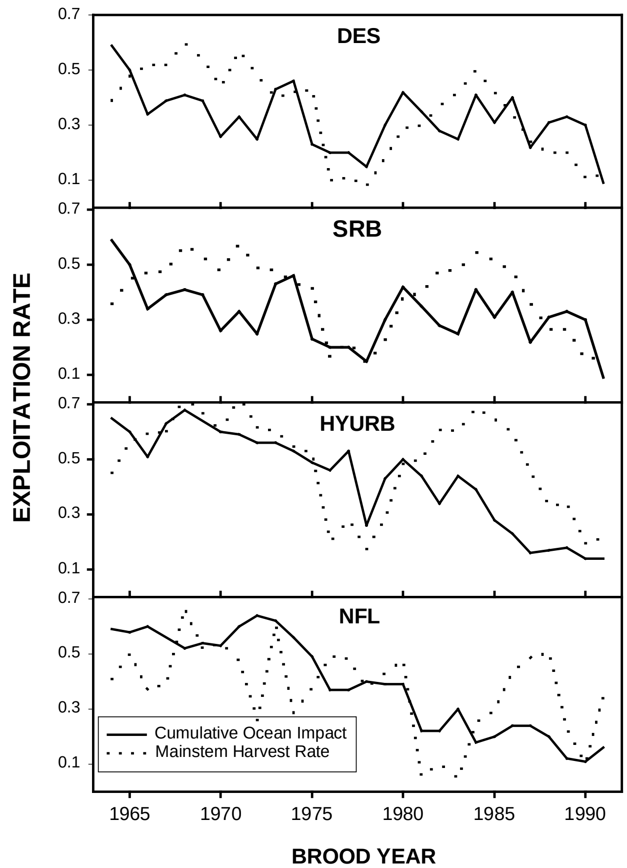

Figure 3.1.2-4: In-river harvest rate of age 4 fish by stock and brood year and cumulative ocean impact through age 4 by stock and brood year, BY 1964-91.

The HYURB population consists of natural origin adult spawners primarily in the Hanford Reach (the last free flowing river reach) of the main stem Columbia River between McNary and Priest Rapids dams, but also includes smaller components like that in the lower Yakima River main stem. The HYURB population migrates a minimum of 540 Rkm past four mainstem Columbia River dams (Figure 3.1.2-2; Table 3.1.2-1). The mainstem reach accessible to spawners is 79 Rkm in length and has the largest cross sectional area compared to the spawning areas of the other three index stocks. The Hanford Reach is presently the primary natural production area for fall chinook in the Columbia River basin. Habitat quality, relative to that used by the other three index stocks, is considered good. Hatchery influences have been variable depending on hatchery production levels at Priest Rapids and Ringold hatcheries and spawning abundance of natural origin adults; an average of 5% of all spawners during 1964-91 BYs were of hatchery origin (Figure 3.1.2-3). Currently, the in-river harvest management regulations are mainly guided by a combination of SRB and URB unit status. The HYURB stock exhibits the highest in-river harvest rate of the four stocks. High cumulative ocean exploitation rates (to age 4) have dropped steadily to below 30% since Brood Year (BY) 1985 (Figure 3.1.2-4), coincident with implementation of the Pacific Salmon Treaty.

The DES population consists of all adult fall chinook salmon spawning in the mainstem Deschutes River between its mouth and Pelton Reregulation Dam near Deschutes Rkm 167. The DES fall chinook population migrates a minimum of 342 Rkm past two mainstem dams of the Columbia River (Figure 3.1.2-2; Table 3.1.2-1). Habitat quality in the Deschutes River is considered fair to good (Table 3.1.2-1).

Spawning habitat exists throughout the river from its mouth to the Pelton Reregulation Dam. No hatchery programs have been implemented for fall chinook salmon in the Deschutes River, and marked hatchery fish have seldomly been seen in creel surveys or at Sherars Falls trap (Rkm 72). Deschutes fall chinook are part of the URB harvest management unit and are subjected to the same in-river harvest management regimes as HYURB stock. Brood year age 4 harvest rates are slightly lower because the DES stock is not subjected to all of the Zone 6 fishery that the HYURB stock experiences. Ocean exploitation rates are the same as SRB (Figure 3.1.2-4).

The NFL population consists of all adult fall chinook salmon spawning downstream of the lower most dam on the North Fork of the Lewis River (Figure 3.1.2-1). The NFL fall chinook population is the only PATH chinook index stock that does not migrate through the Federal Columbia River Power System (FCRPS). Adults migrate a minimum of 160 km from the ocean to the spawning grounds (Table 3.1.2-1). Habitat quality is considered to be fair to good (Table 3.1.2-1). The number of hatchery origin fish observed on spawning grounds is consistently low (Figure 3.1.2-3). The NFL stock is the principle component (>83% on average) of the Columbia River fall chinook management unit known as Lower River Wild (LRW) — bright stocks originating below Bonneville Dam. Other LRW stocks (East Fork Lewis River and Cowlitz River bright stocks in Washington, and Sandy River bright stock in Oregon) are managed as separate stocks within the LRW unit and are not considered linked to NFL productivity. The NFL cumulative ocean exploitation rate has steadily dropped off similar to the HYURB (Figure 3.1.2-4).

Methods

Spawners

For all stocks, spawners are total adult (age 3-6) fish that spawn including both natural and hatchery origin fish.

<u>SRB</u>

The abundance of SRB spawners is estimated from the uppermost Lower Snake River adult dam count: Ice Harbor, 1964-1968; Lower Monumental, 1969; Little Goose, 1970-1974; and Lower Granite, 1975-1996. Adult returning fish enumerated at the Hells Canyon Dam complex (1964-1972) were subtracted from the spawner estimates, because we were only assessing the fish that spawn in presently available habitat. This accounted for recruits that returned from brood years prior to Hells Canyon Dam blocking migration beginning in 1967.

<u>HYURB</u>

The abundance of HYURB spawners is estimated from the McNary to Priest Rapids and Ice Harbor interdam adult count, less McNary pool adult hatchery escapements and harvest for 1964-1997. We concluded that the data available for Yakima River escapement was not sufficient for us to accurately isolate the Hanford Reach component (which would have been preferred).

<u>DES</u>

The DES abundance of spawners above Sherars Falls (RM 43) has been estimated annually since 1977 using Chapman's modification of the Petersen mark-and-recapture method (Ricker 1975; Jonasson and Lindsay 1988). Confederated Tribes of the Warm Springs Indians (CTWSI) and Oregon Department of Fish and Wildlife (ODFW) personnel trap returning salmon at Sherars Falls mark them, and release them above the falls. Marked fish are recovered through carcass surveys conducted upstream of the falls.

The DES escapement estimate below Sherars Falls is determined by multiplying the above-Sherars escapement estimate by the ratio of redds below the falls to redds above the falls. Redd counts are conducted annually by helicopter. Redd counts were not conducted in 1982, 1984, and 1987. For these years, the population estimate is derived by applying an average ratio (of the previous and post run years) of redd counts below the falls to redd counts above the falls.

The total number of adult spawners in the Deschutes River is estimated by adding the above-Sherars Falls to the below-Sherars Falls population estimate and then applying year-specific proportion at age to the population estimate and subtracting the number of jacks from the total population. Age is determined by scale analysis, with the exception of run years 1977 and 1984, where an age-length key based on 1978-83 and 1985-96 scale readings is applied against year-specific length frequencies.

NFL

The run year abundance of spawners is estimated from the multiple-survey peak count of all adult fall chinook carcasses and live fish (bright as well as tule strays), in the 6.4-km index area below Merwin Dam (Rkm 31.4), for 1964-97. Most natural spawning occurs above the Lewis Hatchery (Rkm 25.3), though spawners are found downstream to Rkm 18.5 and the lower reaches of some tributaries (e.g., Cedar Creek). Methods of recovery, counting, and expansion of the index area fish have been consistent since 1964.

Recruits (Freshwater)

For all stocks, adult and jack (age 2 fish) recruits, progeny of the naturally spawning fish, are estimated at the mouth of the Columbia River.

The Columbia River mouth recruits (freshwater) are estimated as follows:

$$Col\,Recruits_{i,by} = \frac{Up\,Recruits_{i,by}}{\left(Conv_{j,by+i} * \left(1 - MainExp_{j,by+i}\right) * \left(1 - TribExp_{j,by+i}\right)\right)}$$

where;

Conv = upstream passage conversion rates
j = jack or adult flag
i= age
by = brood year
yr = return year
MainExp = Columbia River mainstem exploitation rate
TribExp = Tributary exploitation rate
UpRecruits = fish of naturally spawned origin

$$Up\,Recruits_{i,by} = \left(Spawners_{by+i} + TrapFish_{by+i} - HatSpawn_{by+i}\right) * Age\,Prop_{i,by+i}$$

where;

Spawners = total number of spawners
TrapFish = number of natural origin fish removed prior to spawning for artificial production programs
HatSpawn = number of hatchery fish naturally spawning
AgeProp = the proportion of fish at age for a given return year

SRB

Recruits include the uppermost dam counts of natural origin jacks and adults, as well as natural origin fish trapped and removed from Ice Harbor Dam for hatchery broodstock (1977-1993). Recruits removed at Ice Harbor Dam are calculated into Lower Granite Dam equivalents (to account for mortalities expected to occur during upstream passage) by multiplying the number removed at Ice Harbor Dam by the Ice Harbor to Lower Granite dam conversion rate.

The age structure (used to identify wild recruits) for 1986 through 1996 is the Snake River Bright (SRB) year-specific proportion at age reported by the *U.S. v. Oregon* Technical Advisory Committee (TAC 1998). Prior to 1986, the recruit age structure is derived by adjusting the Upriver Bright (URB) year-specific proportion at age (Harlan et al. 1998) by the 1990 - 1997 average URB-SRB age proportion relationship.

Wild recruits above the uppermost Snake River dam are expanded to the mouth of the Columbia River by upstream passage conversion rates (to account for interdam losses including dam mortality) and main stem exploitation rates (Table 3.1.2-2).

Upstream passage conversion rates from Bonneville Dam to the uppermost Snake River dam are calculated by multiplying the Bonneville-McNary rate by the McNary-Ice Harbor rate and the Snake River rates (Table 3.1.2-2). Upstream passage conversion rates from 1980-1996 are for bright fall chinook. Upstream passage conversion rates, prior to the start of the Bonneville Dam Observation (BDO) Program (i.e., pre-1980), are calculated from the upriver run reported in ODFW and WDFW (Washington Department of Fish and Wildlife) (1998). The McNary-Ice Harbor rate is assumed to equal the per dam conversion rate for Bonneville-McNary reach. The Snake rate was calculated from a per dam rate based on Lower Monumental or Little Goose dams as the downriver count. By avoiding Ice Harbor Dam counts, we avoided the bias that would occur as a result of high fall back rates at Ice Harbor (relative to the fall back rate at other Snake dams). The adult conversion rates generated by this method are nearly identical to the conversion rates used in the revised 1996-1998 Biological Assessment of impacts of Columbia River fisheries on listed Snake River salmon (TAC 1998).

Table 3.1.2-2: Subbasin exploitation rate and mainstem conversion and exploitation rates to expand natural SRB escapement to the Snake River area spawning grounds and fisheries, to recruits at the Columbia River mouth. Ocean exploitation rates used to expand Columbia River mouth recruits to account for ocean harvest impacts.

Run Year	Subbasin		Mainstem (Columbia & Snake Rivers)				Ocean Exploitation Rate By Age				
	Exploitation Rate		Conversion Rate		Exploitation Rate						
	Jack	Adult	Jack	Adult	Jack	Adult	2	3	4	5	6
1964	0.000	0.000	1.000	0.383	0.285	0.382					
1965	0.000	0.000	1.000	0.718	0.176	0.519					
1966	0.000	0.000	1.000	0.791	0.076	0.397	0.044				
1967	0.000	0.000	0.947	0.805	0.104	0.499	0.038	0.219			
1968	0.000	0.000	0.627	0.697	0.050	0.358	0.030	0.181	0.447		
1969	0.000	0.000	0.198	0.634	0.065	0.447	0.029	0.141	0.371	0.514	
1970	0.000	0.000	0.250	0.231	0.139	0.472	0.025	0.120	0.210	0.267	0.514
1971	0.000	0.000	0.119	0.207	0.049	0.478	0.025	0.140	0.291	0.345	0.267
1972	0.000	0.000	0.045	0.193	0.056	0.575	0.020	0.136	0.299	0.391	0.345
1973	0.000	0.000	0.073	0.302	0.091	0.530	0.021	0.101	0.279	0.408	0.391
1974	0.000	0.000	0.077	0.094	0.017	0.477	0.014	0.111	0.164	0.205	0.408

Run Year	Subbasin Exploitation Rate		Mainstem (Columbia & Snake Rivers) Conversion Rate		Mainstem Exploitation Rate		Ocean Exploitation Rate By Age				
	Jack	Adult	Jack	Adult	Jack	Adult	2	3	4	5	6
1975	0.000	0.000	0.701	0.293	0.134	0.577	0.027	0.100	0.230	0.329	0.205
1976	0.000	0.000	0.492	0.099	0.067	0.489	0.028	0.147	0.160	0.181	0.329
1977	0.000	0.000	0.609	0.428	0.042	0.480	0.019	0.180	0.317	0.360	0.181
1978	0.000	0.000	0.231	0.391	0.034	0.434	0.015	0.073	0.319	0.402	0.360
1979	0.000	0.000	0.378	0.335	0.021	0.415	0.016	0.082	0.151	0.342	0.402
1980	0.000	0.000	0.348	0.306	0.015	0.161	0.014	0.085	0.115	0.107	0.342
1981	0.000	0.000	0.206	0.238	0.010	0.224	0.014	0.059	0.113	0.163	0.107
1982	0.000	0.000	0.346	0.282	0.012	0.139	0.016	0.107	0.085	0.068	0.163
1983	0.000	0.000	0.425	0.426	0.011	0.226	0.023	0.147	0.202	0.215	0.068
1984	0.000	0.000	0.439	0.911	0.024	0.384	0.025	0.147	0.310	0.357	0.215
1985	0.000	0.000	0.706	0.596	0.067	0.397	0.025	0.105	0.223	0.303	0.357
1986	0.000	0.000	0.524	0.379	0.052	0.482	0.015	0.106	0.170	0.169	0.303
1987	0.000	0.000	0.264	0.376	0.029	0.479	0.037	0.156	0.140	0.159	0.169
1988	0.000	0.000	0.714	0.353	0.044	0.546	0.027	0.060	0.288	0.172	0.159
1989	0.000	0.000	0.538	0.376	0.027	0.515	0.038	0.151	0.233	0.227	0.172
1990	0.000	0.000	0.127	0.378	0.026	0.474	0.042	0.059	0.271	0.252	0.227
1991	0.000	0.000	0.661	0.242	0.051	0.361	0.026	0.051	0.138	0.212	0.252
1992	0.000	0.000	0.206	0.511	0.063	0.266	0.020	0.095	0.242	0.204	0.212
1993	0.000	0.000	0.652	0.560	0.043	0.266	0.006	0.079	0.244	0.204	0.204
1994	0.000	0.000	0.682	0.610	0.031	0.155	0.015	0.014	0.229	0.204	0.204
1995	0.000	0.000	0.348	0.318	0.032	0.171	0.016	0.047	0.074	0.169	0.204
1996	0.000	0.000	0.524	0.367	0.048	0.246		0.046	0.000	0.158	0.169
Mean	0.000	0.000	0.468	0.419	0.060	0.395	0.024	0.108	0.218	0.253	0.257
Min	0.000	0.000	0.045	0.094	0.010	0.139	0.006	0.014	0.000	0.068	0.068
Max	0.000	0.000	1.000	0.911	0.285	0.577	0.044	0.219	0.447	0.514	0.514

During run years 1986 through 1997, mainstem exploitation rates for SRB's are reported by TAC (1998). For the years prior to 1986, the average 1986 through 1992 SRB/URB exploitation rate ratio was used to adjust the HYURB mainstem exploitation rates (Table 3.1.2-3) to derive a SRB mainstem exploitation rate. The 1986 through 1992 years reflect pre-Endangered Species Act harvest management regimes.

Table 3.1.2-3: Subbasin exploitation rate and mainstem conversion and exploitation rates used to expand natural HYURB escapement to the Hanford Reach and Yakima River area spawning grounds and fisheries, to recruits at the Columbia River mouth. Ocean exploitation rates used to expand Columbia River mouth recruits to account for ocean harvest impacts.

Run Year	Subbasin Exploitation Rate		Mainstem (Columbia River) Conversion Rate		Mainstem (Columbia River) Exploitation Rate		Ocean Exploitation Rate by age				
	Jack	Adult	Jack	Adult	Jack	Adult	2	3	4	5	6
1964	0.000	0.000	1.000	0.527	0.359	0.482					
1965	0.000	0.000	1.000	0.802	0.222	0.654					
1966	0.000	0.000	1.000	0.856	0.096	0.500	0.054				
1967	0.000	0.000	0.964	0.865	0.131	0.628	0.049	0.193			
1968	0.000	0.000	0.705	0.763	0.063	0.451	0.042	0.178	0.546		
1969	0.000	0.000	0.379	0.760	0.082	0.564	0.046	0.168	0.485	0.760	
1970	0.000	0.000	0.413	0.425	0.175	0.594	0.044	0.148	0.385	0.652	0.760
1971	0.000	0.000	0.456	0.657	0.062	0.602	0.057	0.213	0.545	0.808	0.652
1972	0.000	0.000	0.294	1.000	0.070	0.725	0.052	0.205	0.569	0.814	0.808
1973	0.000	0.000	0.471	0.812	0.115	0.667	0.049	0.187	0.524	0.774	0.814
1974	0.000	0.000	0.499	0.651	0.022	0.601	0.048	0.172	0.487	0.747	0.774
1975	0.000	0.000	0.838	0.772	0.168	0.727	0.059	0.200	0.481	0.725	0.747
1976	0.000	0.000	0.725	0.460	0.085	0.616	0.050	0.196	0.419	0.642	0.725
1977	0.000	0.000	0.689	0.803	0.053	0.604	0.040	0.162	0.420	0.682	0.642
1978	0.000	0.000	0.333	0.495	0.042	0.546	0.041	0.138	0.411	0.653	0.682
1979	0.000	0.000	0.482	0.560	0.026	0.523	0.069	0.142	0.385	0.584	0.653
1980	0.000	0.000	0.468	0.536	0.019	0.203	0.045	0.191	0.345	0.389	0.584
1981	0.000	0.000	0.306	0.514	0.013	0.282	0.027	0.048	0.371	0.390	0.389
1982	0.000	0.000	0.451	0.515	0.015	0.175	0.033	0.109	0.181	0.234	0.390
1983	0.000	0.000	0.526	0.821	0.014	0.285	0.041	0.096	0.348	0.291	0.234
1984	0.105	0.033	0.539	0.933	0.031	0.484	0.028	0.092	0.424	0.509	0.291
1985	0.150	0.082	0.770	1.000	0.084	0.500	0.039	0.092	0.362	0.447	0.509
1986	0.110	0.038	0.706	1.000	0.066	0.607	0.032	0.126	0.248	0.283	0.447
1987	0.045	0.048	0.696	0.883	0.037	0.604	0.058	0.149	0.333	0.459	0.283
1988	0.080	0.059	0.782	1.000	0.055	0.688	0.054	0.081	0.265	0.400	0.459
1989	0.025	0.064	0.629	0.864	0.034	0.648	0.044	0.070	0.170	0.444	0.400
1990	0.054	0.085	0.651	0.810	0.032	0.598	0.034	0.045	0.125	0.312	0.444
1991	0.064	0.110	0.733	0.751	0.065	0.454	0.023	0.042	0.080	0.274	0.312
1992	0.079	0.057	0.725	0.834	0.080	0.335	0.018	0.040	0.099	0.553	0.274
1993	0.061	0.056	0.725	0.777	0.055	0.335	0.033	0.036	0.129	0.390	0.553
1994	0.075	0.083	0.827	0.802	0.038	0.195	0.015	0.106	0.092	0.310	0.390
1995	0.065	0.083	0.692	0.793	0.040	0.216	0.005	0.011	0.000	0.237	0.310
1996	0.087	0.108	0.616	0.696	0.061	0.310		0.070	0.070	0.104	0.237
Mean	0.030	0.027	0.639	0.750	0.076	0.497	0.041	0.123	0.321	0.495	0.510
Min	0.000	0.000	0.294	0.425	0.013	0.175	0.005	0.011	0.000	0.104	0.234
Max	0.150	0.110	1.000	1.000	0.359	0.727	0.069	0.213	0.569	0.814	0.814

HYURB

Recruits to the spawning ground are the natural origin HYURB spawners. Natural origin (wild) spawners are estimated by subtracting hatchery origin fish that spawn naturally from total number of spawners. The post 1979 hatchery origin spawners were calculated by expanding observed CWT marks on the spawning grounds by sampling and mark rates. In years prior to 1980, only Priest Rapids and Ringold Springs hatcheries had the potential to contribute any significant straying to the spawning grounds. An average spawning ground to hatchery return mark ratio was applied to pre-1980 hatchery returns in order to estimate hatchery contribution to the spawner estimates. The age structure used to identify HYURB wild recruits on spawning grounds is the Upriver Bright (URB) year-specific proportion at age for return years 1964-1997 (Harlan et al. 1998).

Wild recruits on the spawning grounds are expanded to the mouth of the Columbia River by conversion rates (to account for interdam losses including dam mortality) and tributary (subbasin) and mainstem exploitation rates (Table 3.1.2-3). Conversion rates from Bonneville to McNary dams are calculated by dividing McNary Dam count by Bonneville Dam count, less hatchery and tributary escapements and sport and treaty harvests. Conversion rates from 1980-1996 are for bright fall chinook. Conversion rates prior to the Bonneville Dam Observation program are for the upriver run reported in ODFW and WDFW (1998).

The run year-specific jack and adult subbasin exploitation rates for HYURB fall chinook are the significant sport fishery developed on the Hanford Reach, and occasional miscellaneous Tribal fisheries in the NcNary pool. The run year-specific jack and adult mainstem exploitation rates are the sum of lower river harvest and Zone 6 harvest divided by the sum of lower river catch and Bonneville Dam ladder counts. Prior to 1980, the mainstem exploitation rates are for the upriver run reported in ODFW and WDFW (1998).

DES

DES recruits include jacks and adults on the spawning grounds. These recruits are expanded by inriver harvest rates to calculate recruits to the Deschutes River mouth. Deschutes River harvest is estimated by creel censuses of the Indian ceremonial and subsistence fishery at Sherars Falls and the recreational fishery between Sherars Falls and the Deschutes River mouth conducted by CTWSI and ODFW. The number of jacks and adults in the creel is estimated based on length: Jacks are less than 54 cm in length and adults are greater than or equal to 54 cm in length. Annual variations in the number of small fish (< 54 cm) that are actually adults occur in the Deschutes stock. Therefore, in this analysis creel estimates based on length are adjusted using age data, based on scale analysis, to account for small fish that are actually adults.

Recruits by brood year are determined by applying year-specific proportion at age to the population estimates described above. Recruits to the Columbia River mouth are estimated by expanding Deschutes River recruits to account for Columbia River upstream passage conversion rates and recreational, Treaty Indian, and lower Columbia River commercial harvests (Table 3.1.2-4). The conversion rate for Deschutes River fish is defined for this run reconstruction to be the rate for a single lower Columbia interdam reach (Bonneville to The Dalles; Table 3.1.2-4). This rate is the cubed root of the Bonneville to McNary Dam conversion rate.

Table 3.1.2-4: Subbasin exploitation rate and mainstem conversion and exploitation rates used to expand DES natural escapement to the Deschutes River area spawning grounds and fisheries, to recruits at the Columbia River mouth. Ocean exploitation rates used to expand Columbia River mouth recruits to account for ocean harvest impacts.

Run Year	Subbasin Exploitation Rate		Mainstem (partial Columbia River) Conversion Rate		Mainstem (partial Columbia River) Exploitation Rate		Ocean Exploitation Rate (SRB stock) by age				
	Jack	Adult	Jack	Adult	Jack	Adult	2	3	4	5	6
1977	0.445	0.245	0.883	0.930	0.044	0.396	0.019	0.180	0.317	0.360	0.181
1978	0.456	0.342	0.693	0.791	0.038	0.421	0.015	0.073	0.319	0.402	0.360
1979	0.394	0.308	0.784	0.824	0.028	0.426	0.016	0.082	0.151	0.342	0.402
1980	0.416	0.423	0.777	0.812	0.020	0.099	0.014	0.085	0.115	0.107	0.342
1981	0.370	0.319	0.674	0.801	0.015	0.109	0.014	0.059	0.113	0.163	0.107
1982	0.436	0.289	0.767	0.802	0.018	0.080	0.016	0.107	0.085	0.068	0.163
1983	0.489	0.272	0.807	0.936	0.015	0.180	0.023	0.147	0.202	0.215	0.068
1984	0.435	0.291	0.814	0.977	0.026	0.292	0.025	0.147	0.310	0.357	0.215
1985	0.731	0.179	0.917	1.000	0.067	0.300	0.025	0.105	0.223	0.303	0.357
1986	0.509	0.176	0.891	1.000	0.053	0.365	0.015	0.106	0.170	0.169	0.303
1987	0.104	0.252	0.886	0.959	0.036	0.424	0.037	0.156	0.140	0.159	0.169
1988	0.614	0.244	0.921	1.000	0.046	0.497	0.027	0.060	0.288	0.172	0.159
1989	0.481	0.306	0.857	0.952	0.027	0.416	0.038	0.151	0.233	0.227	0.172
1990	0.416	0.295	0.867	0.932	0.021	0.335	0.042	0.059	0.271	0.252	0.227
1991	0.108	0.040	0.902	0.909	0.050	0.242	0.026	0.051	0.138	0.212	0.252
1992	0.004	0.011	0.898	0.941	0.065	0.203	0.020	0.095	0.242	0.204	0.212
1993	0.000	0.002	0.898	0.919	0.038	0.198	0.006	0.079	0.244	0.204	0.204
1994	0.002	0.011	0.939	0.929	0.025	0.114	0.015	0.014	0.229	0.204	0.204
1995	0.004	0.005	0.885	0.925	0.034	0.118	0.016	0.047	0.074	0.169	0.204
1996	0.008	0.010	0.851	0.886	0.046	0.182		0.046	0.000	0.158	0.169
Mean	0.321	0.201	0.845	0.911	0.035	0.270	0.022	0.092	0.193	0.222	0.224
Min	0.000	0.002	0.674	0.791	0.015	0.080	0.006	0.014	0.000	0.068	0.068
Max	0.731	0.423	0.939	1.000	0.067	0.497	0.042	0.180	0.319	0.402	0.402

Harvest rates in both the sport and Treaty fisheries above Bonneville Dam are derived by adjusting Bonneville to McNary (Zone 6) combined commercial and sport harvest rates of bright fall chinook (1980-96) and the upriver run (1977-79) to estimate the rate at which Deschutes River fish are intercepted by Zone 6 commercial gear. Deschutes River fall chinook are not subjected to the same degree of fishing effort as HYURBs, because they do not migrate through the entire Zone 6 fishery (Figure 3.1.2-1). Therefore, the Deschutes River fall chinook harvest rate in the Columbia River is calculated by adjusting the HYURB harvest rates by the proportion of Zone 6 effort that occurs downstream of the mouth of the Deschutes River based on aerial counts of nets conducted annually.

NFL

Hawkins (1998) provides age specific spawning ground escapement estimates for 1964-97. Hatchery strays are subtracted from the total spawners to obtain an estimate of the wild component of the spawning population. Strays are reported in numerous WDFW Columbia River progress reports and memoranda. The strays are estimated by expanding CWT recoveries in the North Fork Lewis by the sampling and mark rates. Four hatcheries consistently represented the majority of strays to the North Fork Lewis River index area during the 1979-1997 return years (when the CWT recovery program was in place). These four WDFW hatcheries were: Lewis Hatchery Complex (Lewis Hatchery and Speelyai Hatchery); Washougal Hatchery; Cowlitz Salmon Hatchery; and the Kalama Hatchery Complex (Fallert Creek [a.k.a., Lower Kalama] Hatchery, and Kalama Falls Hatchery). CWT recoveries in the North Fork Lewis originating from a particular hatchery were compared with the CWT recoveries at that respective hatchery to derive a ratio to account for strays prior to the availability of CWT data.

Wild recruits to the spawning ground are expanded to the mouth of the Columbia River by the North Fork Lewis River tributary (sport) exploitation rates, and by the Columbia River main stem (sport and commercial) exploitation rates for LRW fish (Table 3.1.2-5). Prior to 1980, the exploitation rates are determined from catch and run size data for Lower Columbia River fall chinook (ODFW and WDFW 1998). For 1964-68, average sport exploitation rates were assumed. No applications of upstream passage conversion rates are required to obtain the recruits to the Columbia River mouth because the stock is located below Bonneville Dam.

Table 3.1.2-5: Subbasin and mainstem exploitation rates used to expand NFL natural escapement to the North Fork Lewis River area spawning grounds to recruits at the Columbia River mouth. Ocean exploitation rates used to expand Columbia River mouth recruits to account for ocean harvest impacts.

Run Year	Subbasin Exploitation Rate		Mainstem (L. Columbia) Exploitation Rate		Ocean Exploitation Rate by age				
	Jack	Adult	Jack	Adult	2	3	4	5	6
1964	0.058	0.009	0.143	0.410					
1965	0.058	0.009	0.571	0.502					
1966	0.058	0.009	0.221	0.369	0.051				
1967	0.058	0.009	0.627	0.395	0.053	0.146			
1968	0.058	0.009	0.466	0.666	0.051	0.152	0.494		
1969	0.012	0.012	0.381	0.520	0.051	0.170	0.480	0.340	
1970	0.060	0.010	0.251	0.539	0.044	0.128	0.491	0.310	0.340
1971	0.124	0.009	0.362	0.476	0.050	0.150	0.472	0.516	0.310
1972	0.054	0.013	0.326	0.260	0.045	0.138	0.415	0.221	0.516
1973	0.042	0.005	0.439	0.605	0.049	0.148	0.444	0.311	0.221
1974	0.065	0.007	0.233	0.286	0.053	0.159	0.427	0.265	0.311
1975	0.048	0.005	0.420	0.378	0.068	0.189	0.500	0.287	0.265
1976	0.073	0.004	0.555	0.492	0.066	0.199	0.535	0.301	0.287
1977	0.080	0.005	0.527	0.482	0.060	0.174	0.496	0.336	0.301
1978	0.057	0.005	0.428	0.377	0.045	0.144	0.426	0.263	0.336
1979	0.031	0.003	0.548	0.431	0.058	0.109	0.367	0.229	0.263
1980	0.009	0.031	0.414	0.478	0.042	0.107	0.255	0.173	0.229
1981	0.207	0.044	0.180	0.056	0.027	0.094	0.251	0.100	0.173
1982	0.144	0.070	0.448	0.101	0.017	0.128	0.304	0.232	0.100

Run Year	Subbasin Exploitation Rate		Mainstem (L. Columbia) Exploitation Rate		Ocean Exploitation Rate by age				
	Jack	Adult	Jack	Adult	2	3	4	5	6
1983	0.276	0.077	0.185	0.048	0.010	0.093	0.276	0.306	0.232
1984	0.386	0.098	0.202	0.250	0.011	0.088	0.315	0.220	0.306
1985	0.323	0.076	0.161	0.295	0.011	0.076	0.136	0.179	0.220
1986	0.232	0.061	0.207	0.437	0.012	0.072	0.142	0.216	0.179
1987	0.123	0.045	0.164	0.487	0.019	0.034	0.239	0.237	0.216
1988	0.130	0.041	0.143	0.505	0.021	0.057	0.140	0.240	0.237
1989	0.167	0.078	0.197	0.223	0.022	0.057	0.135	0.192	0.240
1990	0.207	0.059	0.000	0.103	0.037	0.073	0.174	0.302	0.192
1991	0.227	0.081	0.046	0.350	0.020	0.056	0.160	0.191	0.302
1992	0.448	0.136	0.081	0.232	0.015	0.010	0.120	0.211	0.191
1993	0.437	0.170	0.100	0.163	0.014	0.028	0.096	0.231	0.211
1994	0.228	0.073	0.016	0.033	0.003	0.017	0.073	0.204	0.231
1995	0.349	0.195	0.253	0.057	0.000	0.006	0.137	0.159	0.204
1996	0.106	0.007	0.000	0.034	0.003	0.003	0.054	0.143	0.159
Mean	0.150	0.044	0.282	0.334	0.033	0.100	0.295	0.247	0.251
Min	0.009	0.003	0.000	0.033	0.000	0.003	0.054	0.100	0.100
Max	0.448	0.195	0.627	0.666	0.068	0.199	0.535	0.516	0.516

Recruits (Including Ocean Harvest)

For all stocks, adult and jack recruits, progeny of the naturally spawning fish, are estimated at the mouth of the Columbia River and include ocean harvest impacts.

The total recruits are estimated by expanding the Columbia River mouth recruit estimates, from above, by age specific cumulative ocean exploitation rates (see below for definition). The total recruits, which include ocean harvest impacts, are estimated as follows (Deriso 1998):

$$TotRecruits_{i,yr} = \frac{ColRecruits_{i,yr}}{\prod_{j=2}^{i}\left(1 - OCNExp_j\right)}$$

where:

OCNExp = ocean exploitation rate (see below)
i= age
yr = return year
j= the first age fish are vulnerable to ocean fishing

Estimation of Ocean Exploitation Rates

Ocean exploitation rates are estimated from CWT data using the backward cohort method currently used by the Chinook Technical Committee of the Pacific Salmon Commission (PSC-CTC 1988). The exploitation rates are estimated from production groups of CWT subyearling fish from: 1) Lyons Ferry hatchery for SRB stock; 2) Priest Rapids Hatchery and Hanford wild for HYURB stock; 3) North Fork Lewis River wild for NFL; and 4) Lyons Ferry hatchery for DES. The ocean exploitation rates estimated for the hatchery stock groups are used as a surrogate for the natural fall chinook stocks, when natural CWT groups are unavailable (Table 3.1.2-6).

Table 3.1.2-6: Availability of CWT stock groups for estimating ocean exploitation rates.

Natural Fall Stock	Natural CWT Group	Hatchery CWT Group
SRB		Lyons Ferry BY 84-89,91
HYURB	Hanford wild BY 86-91	Priest Rapids BY 75-91
DES	Deschutes BY 77-79 distribution comparison	Lyons Ferry BY 84-89,91
NFL	North Fork Lewis wild BY 77-79,82-91	

The cohort size at any age includes all mortalities which occur in that year plus the number of fish alive at the end of that fishing year (cohort size at age is increased for natural mortality after fishing mortalities have been included). The cohort size is first estimated from the total of all the legal catches and escapement (escapement is adjusted by upstream passage conversion rates for those stocks migrating past dams). Incidental mortalities are then estimated iteratively from the legal catch cohort size and added back into the cohort. Incidental mortalities include sublegal-size fish (shaker) mortalities, and mortalities during chinook non-retention (CNR) fishing seasons. Finally, ocean exploitation rates are calculated as the total ocean fishing mortality (catch + incidental ocean fishery impacts) divided by the cohort size at age less natural mortality.

For brood years when no CWT data are available for the stocks, we used two methods to estimate stock-specific ocean exploitation rates. For brood years 1975 through 1991, when no CWT groups were tagged for a stock of interest, legal catch for each year is estimated using the CWT catch from the closest reference year in the time series where CWT data was available. The CWT legal catch for each fishery in the reference year is adjusted by the ratio of the PSC-CTC fishery index for the reference year to the fishery index of the missing year, for each age in the brood. The CTC fishery index is the ratio of fishery specific exploitation rates in the current year to average fishery specific exploitation rates during the base period (1979-1982), based on CWT data. The basic cohort and exploitation analysis described above is then completed using the estimated legal CWT catch data for these years. The ratio of CWT estimated catch in Columbia River terminal fisheries (all Columbia River net and sport) to the total in-river harvest rate was used to estimate CWT escapement.

For brood years 1964 through 1974, an historic fishery index was first calculated as the ratio of the catch per unit effort in each year to the average catch per unit effort during the base period (catch and effort data were available for the major PSC fisheries and provided by Alaska Department of Fish and Game and Canadian Department of Fisheries and Oceans). The historic fishery index represents the overall effect of changing management regimes relative to a base period (landed catch and effort). The catch for the stock of interest is then estimated by multiplying the average legal catch distribution during the base period (catch years 1979-82, brood years 1975-78) by the historical fishery index for each year. The basic cohort and exploitation analysis described above is then completed for these years before CWT data were available, using historic fishery index adjusted stock catch. Again, the ratio of CWT estimated catch in Columbia River terminal fisheries (net and sport) to the total in-river harvest rate is used to estimate CWT escapement. The historic fishery index approach for estimating stock ocean exploitation rates is similar to the fishery index method used by PSC-CTC (1991).

Wild juvenile Deschutes River fall chinook salmon were only coded-wire tagged in small numbers during BY 1977-79. Ocean exploitation rates for the SRB fall chinook salmon are used as a surrogate for Deschutes River fall chinook ocean exploitation rates because no additional CWT information is available for these fish. Catch distribution of the SRB stock in Pacific Salmon Commission (PSC) ocean fisheries for BY 1984-90 and 1992 is similar to catch distribution of the Deschutes River stock during BY 1977-1979. The proportions of 2- and 3-year-old fish caught in major PSC ocean troll fisheries are more similar between DES and SRB populations than between DES and HYURB or NFL populations. Similarities in patterns of proportion at age, by brood year, between SRB and DES stocks suggest the stocks have similar maturation rates. The Deschutes and SRB brood year age proportions are predominantly 4-year olds, while HYURB and NFL populations are distributed almost equally between both 4- and 5-year old fish. Populations with similar maturation rates and distribution of catch in ocean fisheries are expected to experience comparable ocean exploitation rates. Further, genetic analyses yielded evidence that the SRB and Deschutes populations exhibit similar allele frequencies. These findings have prompted National Marine Fisheries Service (NMFS) to place these stocks in the Snake River Fall-Run evolutionarily significant unit (ESU; Myers et al. 1998). Methods used to derive SRB ocean harvest rates are described above.

Cohort and Exploitation Analysis

Basic steps:

1. calculate initial cohort abundance without incidental mortalities
2. calculate maturation rates at age
3. calculate incidental mortalities (shakers and CNR)
4. re-calculate cohort size
5. expand the cohort size by PSC natural mortality rates (PSC 1988)
6. calculate exploitation rates
7. calculate fishery indices

<u>Cohort Size</u>

$$Cohrt_{by,i} = \frac{Cohrt_{by,i+1} + Esc_{by,i} + \sum_{f=1}^{j}(Cat_{f,i} + shakr_{f,i} + CNRr_{f,i})}{SurvRte_i}$$

where;

Cohrt = cohort size at age
Esc = escapement/upstream passage conversion rate
Cat = catch by fishery at age

shakr = shaker mortalities by fishery by age
CNR = Catch non-retention mortalities by fishery by age
SurvRte = (1-natural mortality by age)
f = fishery
j = all ocean fisheries + Columbia River terminal fisheries
i = age
by = brood year

Ocean Exploitation Rate

$$OCN\exp_i = \frac{TotOcnCat_i}{(Cohrt_i * SurvRte_i)}$$

where:

TotOcnCat = total mortality in all ocean fisheries by age

PSC-CTC Fishery Index

$$CtcFishInd_{yr,f} = \frac{ExpRt_{yr,f}}{ExpRt_{baseavg,f}}$$

where;

CtcFishInd = Pacific Salmon Commission, Chinook Technical Committee, Fishery Index
ExpRt = fishery specific exploitation rate
yr = catch year
baseavg = base period average (years 1979-82)

Historic Fishery Index

$$HistFishIndex_{yr,f} = \frac{CPUE_{yr,f}}{CPUE_{baseavg,f}}$$

where;

HistFishIndex = historic fishery index
CPUE = catch per unit of effort
yr = catch year

Stock Recruitment Analysis

Productivity and survival rate indices were estimated for different periods and fall chinook stocks throughout the Columbia River Basin. Productivity, for a specified time period, is defined as the natural log of the ratio of recruits to spawners in the absence of density dependent mortality (Neave 1953). Productivity is measured here as the intercept, or "a" value, from Ricker (1975):

$$R = e^a S e^{-\beta S}$$

where:

R is recruits and S is Spawners.

The a and β parameters were estimated by the log transformation:

$$\ln(R/S) = a - \beta S$$

Survival rate indices provide a time series of density independent mortality estimates through deviations of observed R/S from those predicted by the fitted stock recruitment function for a specified time period. Survival rate indices were expressed as the natural log of the ratio of observed R/S to the predicted R/S. The natural log of these ratios transforms the differences, such that they tend to be normally distributed.

Ricker production functions were fit to spawner and recruit data for different time periods, and the parameter estimates were compared between time periods, and within and among stocks from different regions of the Columbia River Basin. The Ricker equation was fit to three time periods: pre-1974 brood years (pre lower Snake River dam completion); post-1975 brood years (post lower Snake River dam completion); and all available brood years. While the pre-1975 and post-1974 periods corresponds to the hydro hypotheses on spring/summer chinook (and reflect a 1976 climate regime shift hypothesis), other meaningful periods could have been chosen. Parameter estimates of the Ricker function ("a" and β) were compared to characterize index stock productivity and to test for evidence of density dependence.

Preliminary Results

Within Stock

SRB

Spawner abundance ranged from 246 to 17,655 adults and averaged 3,063 for the period 1964-91 (Table 3.1.2-7). The trend in spawners has exhibited a significant decline since the late 1960s (Figure 3.1.2-5). The total recruitment to the Columbia River mouth ranged from 661 to 57,445 and averaged 14,523 (Table 3.1.2-7). The total recruitment, including ocean harvest, ranged from 714 to 75,389 and averaged 18,668 (Table 3.1.2-8). The natural log of recruits divided by spawners, *ln(R/S)*, ranged from -0.310 to 2.468 and averaged 1.569 for recruitment enumerated at the Columbia River mouth (Table 3.1.2-7). *Ln(R/S)* using recruits harvested in ocean fisheries ranged from -0.232 to 2.904 and averaged 1.830 (Figure 3.1.2-6). The fit of *ln(R/S) vs* spawners yields an intercept (Ricker *a*) of 1.67 using Columbia River mouth recruits and 1.92 including ocean harvest impacts (Tables 3.1.2-9 and 3.1.2-10). The slope (Ricker β) does not change appreciably between fits using recruits without ocean harvest impacts and recruits with ocean harvest impacts (Tables 3.1.2-9 and 3.1.2-10).

Table 3.1.2-7: Run reconstruction for Snake River fall chinook. The recruits do not include cumulative ocean harvest impacts.

Stock	BY	S_{adult}	R_2	R_3	R_4	R_5	R_6	R_{adult}	R_{total}	R_{obs}/S	Ln(R/S)
SRB	1964	7682	2514	6993	11838	66	0	18896	21410	2.787	1.025
SRB	1965	7011	6930	29074	7172	1187	0	37433	44362	6.327	1.845
SRB	1966	8591	9255	7292	12365	660	0	20318	29573	3.442	1.236
SRB	1967	11385	13304	24168	19555	418	0	44141	57445	5.045	1.618
SRB	1968	17655	7706	19490	9867	225	0	29582	37288	2.112	0.748
SRB	1969	4836	15804	9054	4508	583	0	14145	29950	6.193	1.823
SRB	1970	4416	17319	11403	10713	863	0	22979	40297	9.126	2.211

Stock	BY	S_{adult}	R_2	R_3	R_4	R_5	R_6	R_{adult}	R_{total}	R_{obs}/S	Ln(R/S)
SRB	1971	4241	12992	3866	5269	798	0	9932	22924	5.405	1.687
SRB	1972	1437	6989	5620	4144	188	0	9952	16942	11.793	2.468
SRB	1973	2208	1221	9243	2574	294	0	12111	13333	6.038	1.798
SRB	1974	741	1265	3037	3799	505	0	7341	8606	11.614	2.452
SRB	1975	1450	1432	1704	4726	835	103	7368	8800	6.068	1.803
SRB	1976	706	2562	993	1518	620	18	3149	5711	8.090	2.091
SRB	1977	1059	1458	891	4827	493	0	6211	7669	7.243	1.980
SRB	1978	918	656	1175	2180	172	43	3569	4226	4.605	1.527
SRB	1979	901	3605	3800	1860	93	8	5760	9365	10.395	2.341
SRB	1980	554	2829	1592	923	373	79	2968	5797	10.462	2.348
SRB	1981	999	1234	622	1303	368	58	2350	3584	3.589	1.278
SRB	1982	1252	1675	1515	3071	409	37	5032	6708	5.358	1.679
SRB	1983	960	1499	2702	2007	191	141	5041	6539	6.810	1.918
SRB	1984	778	1680	1360	744	1381	12	3497	5178	6.653	1.895
SRB	1985	1148	543	190	1252	324	68	1834	2377	2.070	0.727
SRB	1986	1475	70	838	850	783	20	2490	2560	1.736	0.551
SRB	1987	1094	227	270	1054	241	4	1568	1795	1.642	0.496
SRB	1988	246	1247	629	715	290	20	1654	2901	11.771	2.466
SRB	1989	832	47	422	1273	237	1	1933	1980	2.380	0.867
SRB	1990	284	498	157	274	90	80	601	1100	3.875	1.354
SRB	1991	901	108	188	306	59	0	552	661	0.733	-0.310
SRB	1992	632									
SRB	1993	901									
SRB	1994	549									
SRB	1995	334									
SRB	1996	1079									
SRB	1997	1007									
	Ave	2655	4167	5296	4310	455	25	10086	14253	5.834	1.569
	Min	246	47	157	274	59	0	552	661	0.733	-0.310
	Max	17655	17319	29074	19555	1381	141	44141	57445	11.793	2.468

Table 3.1.2-8: Run reconstruction for Snake River fall chinook. The recruits include cumulative ocean harvest impacts.

Stock	BY	S_{adult}	R_2	R_3	R_4	R_5	R_6	R_{adult}	R_{total}	R_{obs}/S	Ln(R/S)
SRB	1964	7682	2631	9367	28666	327	0	38361	40991	5.336	1.674
SRB	1965	7011	7201	36898	14461	3265	0	54624	61825	8.818	2.177
SRB	1966	8591	9546	8754	18784	1532	0	29070	38617	4.495	1.503
SRB	1967	11385	13697	28283	32276	1133	0	61692	75389	6.622	1.890
SRB	1968	17655	7902	23244	16790	647	0	40682	48584	2.752	1.012
SRB	1969	4836	16209	10748	7421	1209	0	19377	35586	7.359	1.996
SRB	1970	4416	17666	12940	14546	1746	0	29232	46898	10.620	2.363
SRB	1971	4241	13272	4440	7858	1452	0	13750	27022	6.371	1.852
SRB	1972	1437	7088	6329	5557	395	0	12281	19369	13.483	2.601
SRB	1973	2208	1256	11137	4542	869	0	16548	17804	8.063	2.087
SRB	1974	741	1302	3812	6998	1413	0	12223	13525	18.252	2.904
SRB	1975	1450	1459	1874	6118	1209	167	9368	10827	7.466	2.010
SRB	1976	706	2602	1098	1897	926	32	3954	6556	9.288	2.229
SRB	1977	1059	1482	990	6047	662	0	7699	9181	8.671	2.160
SRB	1978	918	666	1265	2565	257	83	4170	4836	5.270	1.662
SRB	1979	901	3658	4317	2646	205	27	7195	10853	12.046	2.489
SRB	1980	554	2876	1898	1594	925	282	4698	7575	13.670	2.615
SRB	1981	999	1263	746	2009	683	129	3567	4830	4.836	1.576
SRB	1982	1252	1719	1737	4241	672	73	6723	8442	6.743	1.909
SRB	1983	960	1536	3098	2677	307	275	6357	7894	8.221	2.107
SRB	1984	778	1706	1636	1258	3019	34	5947	7653	9.833	2.286
SRB	1985	1148	564	210	1804	624	175	2813	3377	2.940	1.079
SRB	1986	1475	72	1014	1411	1650	53	4129	4201	2.848	1.047

Stock	BY	S_{adult}	R_2	R_3	R_4	R_5	R_6	R_{adult}	R_{total}	R_{obs}/S	Ln(R/S)
SRB	1987	1094	236	298	1351	388	7	2044	2281	2.086	0.735
SRB	1988	246	1301	692	1037	529	45	2302	3603	14.622	2.683
SRB	1989	832	48	479	1911	447	2	2839	2887	3.471	1.244
SRB	1990	284	508	174	394	155	166	890	1398	4.925	1.594
SRB	1991	901	109	192	337	77		605	714	0.793	-0.232
	Ave	3063	4271	6345	7043	954	57	14398	18668	7.496	1.830
	Min	246	48	174	337	77	0	605	714	0.793	-0.232
	Max	17655	17666	36898	32276	3265	282	61692	75389	18.252	2.904

Table 3.1.2-9: Ricker spawner-recruit function parameters, fits, predicted spawners at key recruitment levels. Note, function fit to recruits at Columbia River mouth (*without* ocean harvest impacts).

Statistic	Period	Pre 1975 (BY 64-74)				Post 1974 (BY 75-91)				All Years (BY 64-91)			
	Stock	NFL	DES	HYURB	SRB	NFL	DES	HYURB	SRB	NFL	DES	HYURB	SRB
a		1.445	n/a	2.270	2.311	2.220	2.583	2.138	2.237	1.888	2.583	2.187	1.673
Coeff. of Variation		*29.7%*	n/a	*38.7%*	*7.1%*	*15.2%*	*16.3%*	*17.2%*	*23.7%*	*14.3%*	*16.3%*	*10.6%*	*9.9%*
$\alpha = e^a$		*4.243*	n/a	*9.682*	*10.080*	*9.208*	*13.231*	*8.482*	*9.369*	*6.608*	*13.231*	*8.906*	*5.325*
β		0.00008	n/a	0.00002	0.00009	0.00016	0.00038	0.00002	0.00084	0.00013	0.00038	0.00002	0.00003
Coeff. of Variation		*43.5%*	n/a	*133.4%*	*22.4%*	*16.9%*	*22.6%*	*29.3%*	*65.0%*	*17.2%*	*22.6%*	*21.6%*	*97.4%*
Prob.($\beta \le 0$)		*0.0235*	n/a	*0.2365*	*0.0008*	*0.0000*	*0.0003*	*0.0019*	*0.0723*	*0.0000*	*0.0003*	*0.0000*	*0.1567*
S_{msy}		6,920	n/a	31,293	8,436	4,789	2,178	33,859	918	5,405	2,178	33,400	18,895
S_{msp}		12,005	n/a	40,411	10,793	6,260	2,642	45,204	1,194	7,782	2,642	44,027	29,503
S_{rep}		17,351	n/a	91,747	24,938	13,897	6,824	96,645	2,673	14,695	6,824	96,274	49,344
r^2		0.37	n/a	0.06	0.69	0.70	0.60	0.44	0.14	0.57	0.60	0.45	0.04
# observations		11	0	11	11	17	15	17	17	28	15	28	28
Spawner Range:													
minimum obsv.		4,130	n/a	19,327	741	3,371	2,320	14,213	246	3,371	2,320	14,213	246
mean obsv.		10,680	n/a	26,202	6,382	11,442	4,683	49,209	915	11,143	4,683	40,171	3,063
maximum obsv.		19,926	n/a	36,343	17,655	21,199	7,903	105,347	1,475	21,199	7,903	105,347	17,655

Table 3.1.2-10: Ricker spawner-recruit function parameters, fits, predicted spawners at key recruitment levels. Note, function fit to recruits *with* ocean harvest impacts.

Statistic	Period	Pre 1975 (BY 64-74)				Post 1974 (BY 75-91)				All Years (BY 64-91)			
	Stock	NFL	DES	HYURB	SRB	NFL	DES	HYURB	SRB	NFL	DES	HYURB	SRB
a		2.148	n/a	3.365	2.580	2.696	2.843	2.654	2.418	2.479	2.843	2.960	1.919
Coeff. of Variation		*17.8%*	n/a	*26.5%*	*5.6%*	*12.7%*	*15.9%*	*15.9%*	*21.6%*	*11.8%*	*15.9%*	*9.0%*	*8.5%*
$\alpha = e^a$		*8.564*	n/a	*28.947*	*13.194*	*14.813*	*17.161*	*14.210*	*11.220*	*11.935*	*17.161*	*19.290*	*6.812*
β		0.00008	n/a	0.00004	0.00009	0.00017	0.00037	0.00002	0.00076	0.00014	0.00037	0.00003	0.00003
Coeff. of Variation		*42.2%*	n/a	*94.2%*	*20.4%*	*16.0%*	*24.7%*	*31.2%*	*70.5%*	*17.7%*	*24.7%*	*20.6%*	*112.5%*
Prob.($\beta \le 0$)		*0.0209*	n/a	*0.1582*	*0.0004*	*0.0000*	*0.0007*	*0.0029*	*0.0881*	*0.0000*	*0.0007*	*0.0000*	*0.1912*
S_{msy}		9,844	n/a	25,026	9,157	4,890	2,302	35,118	1,052	5,967	2,302	31,602	24,351
S_{msp}		13,108	n/a	28,122	11,113	5,827	2,691	42,112	1,315	7,372	2,691	36,464	34,705
S_{rep}		28,152	n/a	94,644	28,668	15,707	7,649	111,761	3,180	18,279	7,649	107,919	66,588
r^2		0.38	n/a	0.11	0.73	0.72	0.56	0.41	0.12	0.55	0.56	0.48	0.03
# observations		11	0	11	11	17	15	17	17	28	15	28	28
Spawner Range:													
minimum obsv.		4,130	n/a	19,327	741	3,371	2,320	14,213	246	3,371	2,320	14,213	246
mean obsv.		10,680	n/a	26,202	6,382	11,442	4,683	49,209	915	11,143	4,683	40,171	3,063
maximum obsv.		19,926	n/a	36,343	17,655	21,199	7,903	105,347	1,475	21,199	7,903	105,347	17,655

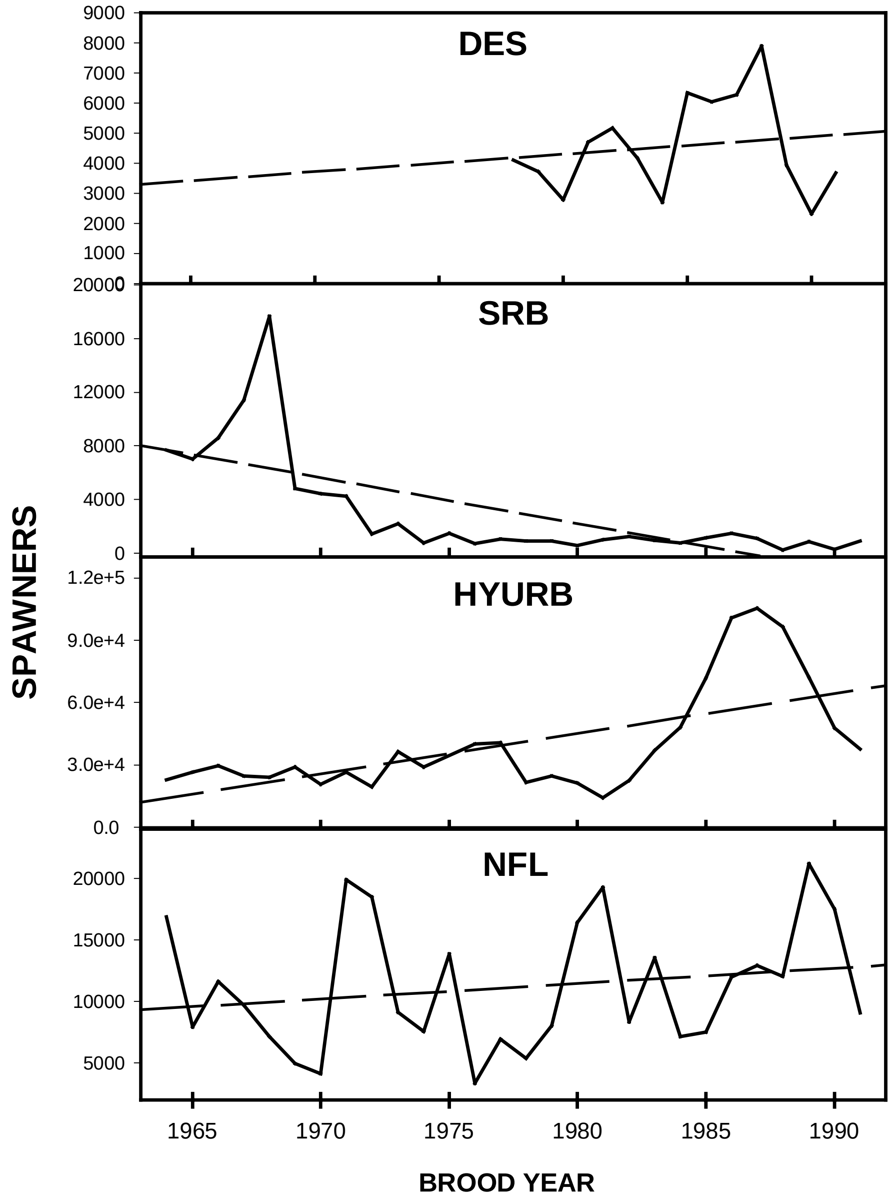

Figure 3.1.2-5: Spawner abundance of four fall chinook stocks, BY 1964-91. Trend line is represented by the dashed line.

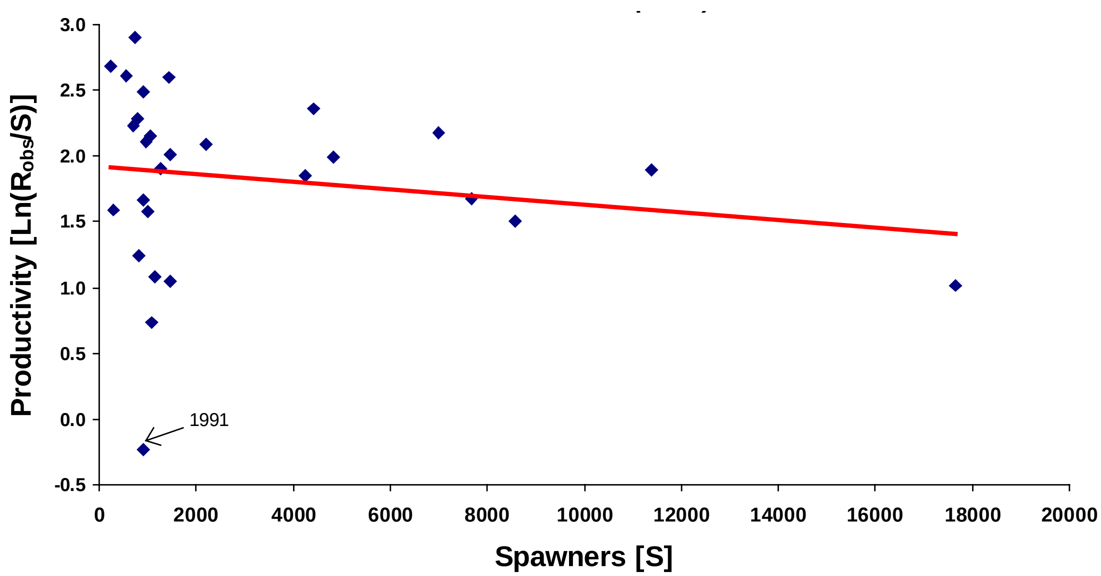

Figure 3.1.2-6: Productivity versus spawners for Snake River fall chinook (BY 1964-1991). Recruits includes cumulative ocean impacts.

<u>HYURB</u>

Spawner abundance ranged from 14,213 to 105,347 adults and averaged 40,171 for the period 1964-91 (Table 3.1.2-11). HYURB spawner abundance has exhibited a significant increasing trend for the time series (Figure 3.1.2-5). Total recruitment to the Columbia River mouth ranged from 32,798 to 506,351 and averaged 149,734 (Table 3.1.2-11). Total recruitment including ocean harvest ranged from 38,067 to 956,878 and averaged 282,685 (Table 3.1.2-12). The natural log of recruits divided by spawners, *ln(R/S)*, ranged from -0.509 to 2.517 and averaged 1.274 for recruitment enumerated at the Columbia River mouth (Tables 3.1.2-9 and 3.1.2-10). *Ln(R/S)* using recruits harvested in ocean fisheries ranged from -0.041 to 3.359 and averaged 1.858 (Figure 3.1.2-7). A fit of *ln(R/S) vs* spawners yields a Ricker "a" of 2.19 using Columbia River mouth recruits and 2.96 including ocean harvest impacts (Tables 3.1.2-9 and 3.1.2-10). The Ricker "beta" does not change appreciably between plots using recruits without ocean harvest impacts and recruits with ocean harvest impacts (Tables 3.1.2-9 and 3.1.2-10).

Table 3.1.2-11: Run reconstruction for Hanford Reach/Yakima River upriver bright fall chinook. The recruits do not include cumulative ocean harvest impacts.

Stock	BY	S_{adult}	R_2	R_3	R_4	R_5	R_6	R_{adult}	R_{total}	R_{obs}/S	Ln(R/S)
HYURB	1964	22703	10186	13383	20635	1018	0	35037	45222	1.992	0.689
HYURB	1965	26668	14269	35113	48168	8444	0	91725	105994	3.975	1.380
HYURB	1966	29724	12586	35390	45277	4085	0	84752	97338	3.275	1.186
HYURB	1967	24638	43604	53882	52392	3710	0	109984	153588	6.234	1.830
HYURB	1968	24035	23683	36234	39545	5028	0	80807	104490	4.347	1.470
HYURB	1969	28937	17939	25263	45662	9580	0	80504	98443	3.402	1.224
HYURB	1970	20511	32326	79738	77108	28013	0	184859	217185	10.589	2.360
HYURB	1971	26393	36240	19307	77253	32214	0	128773	165013	6.252	1.833
HYURB	1972	19327	39549	57181	75556	10414	0	143151	182700	9.453	2.246
HYURB	1973	36343	24405	116946	64085	10944	0	191975	216380	5.954	1.784
HYURB	1974	28940	50625	52470	63566	15379	0	131415	182040	6.290	1.839
HYURB	1975	34628	50267	19788	64833	19650	231	104503	154769	4.470	1.497
HYURB	1976	39987	38333	9449	18663	7314	67	35493	73826	1.846	0.613
HYURB	1977	40745	26577	6732	24136	7029	0	37896	64474	1.582	0.459
HYURB	1978	21644	13463	3829	19294	4891	1144	29158	42620	1.969	0.678
HYURB	1979	24840	27256	23265	34711	12942	146	71064	98320	3.958	1.376
HYURB	1980	21224	39286	15212	54481	35863	1457	107013	146299	6.893	1.931
HYURB	1981	14213	41500	26295	57181	32843	1862	118181	159682	11.235	2.419
HYURB	1982	22598	73461	47845	94786	60274	3635	206540	280001	12.390	2.517
HYURB	1983	37038	90744	69850	130314	74916	2816	277896	368640	9.953	2.298
HYURB	1984	48149	109856	65223	177651	147196	6426	396495	506351	10.516	2.353
HYURB	1985	71732	27508	25173	56578	61989	3979	147719	175227	2.443	0.893
HYURB	1986	100626	21037	25621	65634	56168	949	148372	169409	1.684	0.521
HYURB	1987	105347	14764	8145	21919	18145	371	48581	63345	0.601	-0.509
HYURB	1988	96329	18088	8786	26864	17948	190	53789	71877	0.746	-0.293
HYURB	1989	72022	21254	12979	41501	34985	1148	90612	111866	1.553	0.440
HYURB	1990	47856	19760	8889	39551	35953	511	84905	104665	2.187	0.783
HYURB	1991	37580	7457	8219	9418	7704	0	25341	32798	0.873	-0.136
HYURB	1992	34371									
HYURB	1993	35322									
HYURB	1994	54373									
HYURB	1995	39936									
HYURB	1996	38443									
HYURB	1997	37685									
	Ave	40144	33787	32507	55241	27309	890	115948	149734	4.881	1.274
	Min	14213	7457	3829	9418	1018	0	25341	32798	0.601	-0.509
	Max	105347	109856	116946	177651	147196	6426	396495	506351	12.390	2.517

Table 3.1.2-12: Run reconstruction for Hanford Reach/Yakima River upriver bright fall chinook. The recruits include cumulative ocean harvest impacts.

Stock	BY	S_{adult}	R_2	R_3	R_4	R_5	R_6	R_{adult}	R_{total}	R_{obs}/S	Ln(R/S)
HYURB	1964	22703	10763	17513	59544	12223	0	89280	100043	4.407	1.483
HYURB	1965	26668	15010	44914	119584	60173	0	224670	239681	8.988	2.196
HYURB	1966	29724	13143	44392	92326	43371	0	180088	193231	6.501	1.872
HYURB	1967	24638	45720	66291	141592	53869	0	261752	307471	12.480	2.524
HYURB	1968	24035	24786	48195	122057	68632	0	238885	263670	10.970	2.395
HYURB	1969	28937	19020	33688	127808	105812	0	267309	286328	9.895	2.292
HYURB	1970	20511	34099	103499	195157	257375	0	556032	590130	28.771	3.359
HYURB	1971	26393	38105	24531	189054	219932	0	433517	471622	17.870	2.883
HYURB	1972	19327	41534	75064	170666	73927	0	319656	361190	18.689	2.928
HYURB	1973	36343	25922	154435	145947	71908	0	372290	398212	10.957	2.394
HYURB	1974	28940	53291	65896	135570	78823	0	280289	333580	11.526	2.445
HYURB	1975	34628	52384	23924	127382	63229	1218	215752	268136	7.743	2.047

Stock	BY	S_{adult}	R_2	R_3	R_4	R_5	R_6	R_{adult}	R_{total}	R_{obs}/S	Ln(R/S)
HYURB	1976	39987	39951	11477	34599	22220	335	68630	108581	2.715	0.999
HYURB	1977	40745	28539	8939	50979	19370	0	79288	107827	2.646	0.973
HYURB	1978	21644	14097	4211	25922	9272	3061	42466	56563	2.613	0.961
HYURB	1979	24840	28023	26848	61409	46673	1073	136003	164027	6.603	1.888
HYURB	1980	21224	40623	17400	108258	128930	9474	264063	304686	14.356	2.664
HYURB	1981	14213	43294	30215	102929	82474	6524	222142	265436	18.675	2.927
HYURB	1982	22598	75565	54181	142643	167796	18720	383340	458905	20.307	3.011
HYURB	1983	37038	94407	83136	232648	222884	13963	552631	647038	17.470	2.860
HYURB	1984	48149	113468	79132	293205	436790	34283	843410	956878	19.873	2.989
HYURB	1985	71732	29205	29094	78826	125482	11701	245103	274308	3.824	1.341
HYURB	1986	100626	22243	29125	85270	100551	2341	217286	239529	2.380	0.867
HYURB	1987	105347	15441	8923	26107	48397	2218	85646	101086	0.960	-0.041
HYURB	1988	96329	18722	9496	32235	35324	614	77669	96391	1.001	0.001
HYURB	1989	72022	21752	13830	50762	61993	2946	129532	151284	2.101	0.742
HYURB	1990	47856	20120	9386	45968	54777	1021	111151	131271	2.743	1.009
HYURB	1991	37580	7712	9508	10895	9951		30355	38067	1.013	0.013
	Ave	40171	35248	40259	107477	95791	4055	247437	282685	9.574	1.858
	Min	14213	7712	4211	10895	9272	0	30355	38067	0.960	-0.041
	Max	105347	113468	154435	293205	436790	34283	843410	956878	28.771	3.359

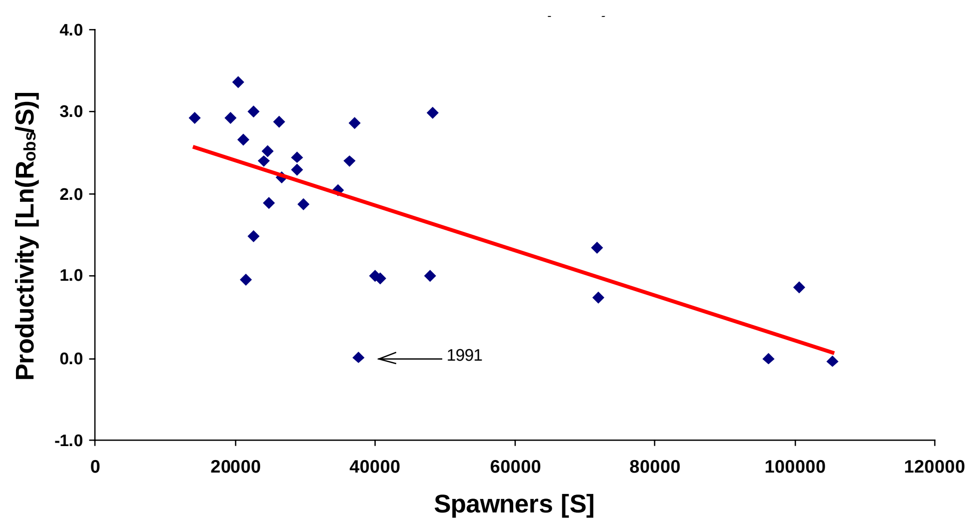

Figure 3.1.2-7: Productivity versus spawners for Hanford Reach/Yakima River fall chinook (BY 1964-1991). Recruits include ocean harvest impacts).

<u>DES</u>

Spawner abundance ranged from 2,320 to 7,903 adults and averaged 4,683 for the period 1977-91 (Table 3.1.2-13). DES spawner abundance has not exhibited a significant trend over the period 1977-91 (Figure 3.1.2-5). Total recruitment to the Columbia River mouth ranged from 3,261 to 35,515 and averaged 11,558 (Table 3.1.2-13). Total recruitment including ocean harvest ranged from 4,125 to 56,348 and averaged 15,810 (Table 3.1.2-14). The natural log of recruits divided by spawners, *ln(R/S)*, ranged from -0.655 to 2.580 and averaged 0.810 for recruitment enumerated at the Columbia River mouth (Tables 3.1.2-9 and 3.1.2-10). *Ln(R/S)* using recruits harvested in ocean fisheries ranged from -0.420 to 3.042 and averaged 1.102 (Figure 3.1.2-8). A fit of *ln(R/S) vs* spawners yields a Ricker "a" of 2.58 using Columbia River mouth recruits and 2.84 including ocean harvest impacts (Tables 3.1.2-9 and 3.1.2-10). The Ricker "beta" does not change between plots using recruits without ocean harvest impacts and recruits with ocean harvest impacts (Tables 3.1.2-9 and 3.1.2-10).

Table 3.1.2-13: Run reconstruction for Deschutes River fall chinook. The recruits do not include cumulative ocean harvest impacts.

Stock	BY	S_{adult}	R_2	R_3	R_4	R_5	R_6	R_{adult}	R_{total}	R_{obs}/S	Ln(R/S)
DES	1977	6414	6437	3481	5008	716	0	9205	15642	2.439	0.891
DES	1978	4099	4456	4044	4775	1114	0	9932	14388	3.510	1.256
DES	1979	3728	3799	4377	3921	645	0	8943	12743	3.418	1.229
DES	1980	2788	4323	2406	2742	1411	0	6560	10883	3.904	1.362
DES	1981	4704	1595	2097	6775	1382	0	10254	11848	2.519	0.924
DES	1982	5176	2475	2823	6156	886	0	9865	12340	2.384	0.869
DES	1983	4160	1411	4020	5044	1868	150	11083	12494	3.003	1.100
DES	1984	2690	3011	9271	17937	5085	211	32503	35515	13.201	2.580
DES	1985	6333	2022	972	4337	1336	0	6645	8668	1.369	0.314
DES	1986	6045	167	598	3165	2499	54	6316	6484	1.073	0.070
DES	1987	6278	374	598	1922	299	68	2887	3261	0.519	-0.655
DES	1988	7903	396	1153	2448	1964	0	5565	5962	0.754	-0.282
DES	1989	3927	639	1850	3454	1097	0	6400	7039	1.793	0.584
DES	1990	2320	1424	2844	4661	1755	30	9290	10714	4.618	1.530
DES	1991	3684	2017	1645	1277	447	0	3368	5385	1.462	0.380
DES	1992	3454									
DES	1993	6126									
DES	1994	6025									
DES	1995	6603									
DES	1996	7734									
DES	1997	17618									
	Ave	5610	2303	2812	4908	1500	34	9254	11558	3.064	0.810
	Min	2320	167	598	1277	299	0	2887	3261	0.519	-0.655
	Max	7903	6437	9271	17937	5085	211	32503	35515	13.201	2.580

Table 3.1.2-14: Run reconstruction for Deschutes River fall chinook. The recruits include cumulative ocean harvest impacts.

Stock	BY	S_{adult}	R_2	R_3	R_4	R_5	R_6	R_{adult}	R_{total}	R_{obs}/S	Ln(R/S)
DES	1977	6414	6542	3863	6273	963	0	11099	17641	2.750	1.012
DES	1978	4099	4518	4366	5618	1671	0	11654	16172	3.945	1.372
DES	1979	3728	3855	4969	5580	1427	0	11976	15831	4.247	1.446
DES	1980	2788	4395	2862	4735	3498	0	11095	15490	5.556	1.715
DES	1981	4704	1632	2498	10450	2566	0	15513	17145	3.645	1.293
DES	1982	5176	2540	3228	8503	1455	0	13186	15725	3.038	1.111
DES	1983	4160	1447	4614	6728	3010	291	14643	16090	3.867	1.353
DES	1984	2690	3058	11259	30319	11116	596	53290	56348	20.944	3.042
DES	1985	6333	2101	1050	6250	2573	0	9873	11974	1.891	0.637
DES	1986	6045	172	732	5257	5270	146	11404	11576	1.915	0.650
DES	1987	6278	389	653	2464	482	137	3737	4125	0.657	-0.420
DES	1988	7903	414	1263	3550	3578	0	8391	8804	1.114	0.108
DES	1989	3927	656	2134	5184	2069	0	9387	10043	2.558	0.939
DES	1990	2320	1453	3170	6694	3033	66	12963	14416	6.213	1.827
DES	1991	3684	2030	1701	1407	627	0	3735	5765	1.565	0.448
	Ave	4683	2347	3224	7267	2889	82	13463	15810	4.260	1.102
	Min	2320	172	653	1407	482	0	3735	4125	0.657	-0.420
	Max	7903	6542	11259	30319	11116	596	53290	56348	20.944	3.042

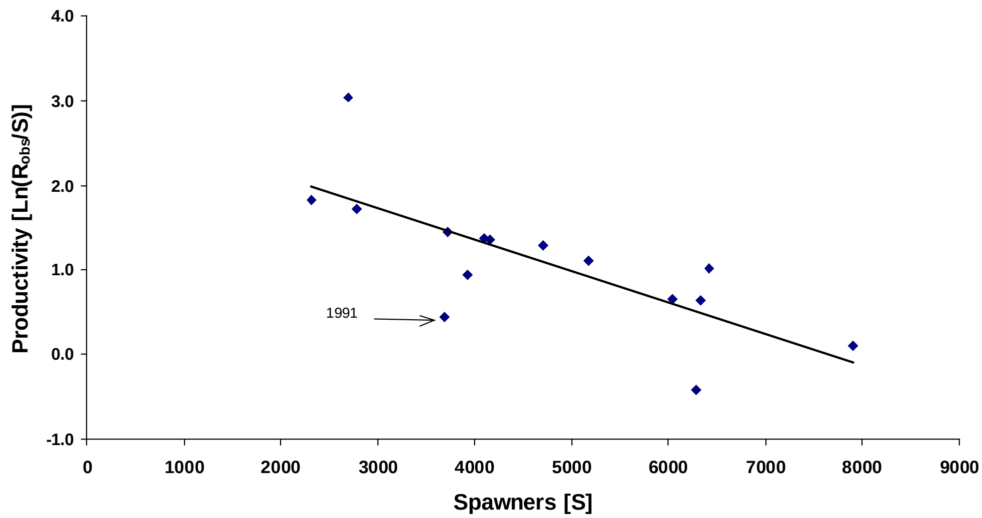

Figure 3.1.2-8: Productivity versus spawners for Deschutes River fall chinook (BY 1977-91). Recruits include cumulative ocean harvest impacts.

NFL

Spawner abundance ranged from 3,371 to 21,199 adults and averaged 11,143 for the period 1964-91 (Table 3.1.2-15). NFL spawner abundance has not exhibited a significant trend over the period 1964-91 (Figure 3.1.2-5). Total recruitment to the Columbia River mouth ranged from 2,351 to 56,551 and averaged 18,554 (Table 3.1.2-15). Total recruitment including ocean harvest ranged from 2,591 to 91,246 and averaged 31,396 (Table 3.1.2-16). The natural log of recruits divided by spawners, *ln(R/S)*, ranged from -2.199 to 2.067 and averaged 0.456 for recruitment enumerated at the Columbia River mouth (Table 3.1.2-15). *Ln(R/S)* using recruits harvested in ocean fisheries ranged from -2.102 to 2.545 and averaged 0.968 (Figure 3.1.2-9). A fit of *ln(R/S) vs* spawners yields a Ricker "a" of 1.89 using Columbia River mouth recruits and 2.48 including ocean harvest impacts (Tables 3.1.2-15 and 3.1.2-16). The Ricker "beta" does not change between plots using recruits without ocean harvest impacts and recruits with ocean harvest impacts (Tables 3.1.2-9 and 3.1.2-10).

Table 3.1.2-15: Run reconstruction for North Fork Lewis River naturally spawning fall chinook. The recruits do not account for cumulative ocean harvest impacts.

Stock	BY	S_{adult}	R_2	R_3	R_4	R_5	R_6	R_{adult}	R_{total}	R_{obs}/S	Ln(R/S)
NFL	1964	16857	2023	2993	14580	4049	0	21622	23646	1.403	0.338
NFL	1965	7927	1053	1698	3835	918	0	6451	7504	0.947	-0.055
NFL	1966	11627	363	1683	4529	4983	0	11195	11557	0.994	-0.006
NFL	1967	9711	1289	2184	22416	2814	0	27413	28702	2.956	1.084
NFL	1968	7160	24990	7429	18649	5482	0	31561	56551	7.898	2.067
NFL	1969	4986	857	2425	10779	2395	0	15598	16455	3.300	1.194
NFL	1970	4130	1095	3483	4662	4783	0	12928	14024	3.396	1.222
NFL	1971	19926	7247	2651	12209	1382	0	16242	23489	1.179	0.165
NFL	1972	18488	2469	3905	2978	2274	0	9157	11626	0.629	-0.464
NFL	1973	9120	1892	1219	6421	1404	0	9045	10937	1.199	0.182
NFL	1974	7549	2007	3459	3742	1890	0	9091	11098	1.470	0.385
NFL	1975	13859	1944	2251	7679	3135	0	13064	15008	1.083	0.080
NFL	1976	3371	1380	2761	17879	2237	20	22898	24278	7.202	1.974
NFL	1977	6930	2125	5182	14725	2771	0	22677	24803	3.579	1.275
NFL	1978	5363	272	1899	2815	2811	141	7666	7938	1.480	0.392
NFL	1979	8023	3017	2972	8995	3138	25	15129	18146	2.262	0.816

Stock	BY	S_{adult}	R_2	R_3	R_4	R_5	R_6	R_{adult}	R_{total}	R_{obs}/S	Ln(R/S)
NFL	1980	16394	1720	1458	5018	1728	125	8329	10048	0.613	-0.489
NFL	1981	19297	2042	1748	5377	2627	29	9780	11822	0.613	-0.490
NFL	1982	8370	1868	2697	9667	5025	0	17389	19257	2.301	0.833
NFL	1983	13540	3473	5755	11124	7261	223	24364	27837	2.056	0.721
NFL	1984	7132	4174	6389	10796	15058	1785	34028	38202	5.356	1.678
NFL	1985	7491	5651	3608	8336	8528	655	21128	26779	3.575	1.274
NFL	1986	11983	3481	3165	6544	7281	485	17476	20956	1.749	0.559
NFL	1987	12935	2669	553	4337	2778	446	8115	10784	0.834	-0.182
NFL	1988	12059	1810	1931	3786	4765	397	10879	12689	1.052	0.051
NFL	1989	21199	770	636	745	167	33	1581	2351	0.111	-2.199
NFL	1990	17506	3255	2821	6871	9383	629	19703	22957	1.311	0.271
NFL	1991	9066	1246	1364	3076	4326	47	8813	10059	1.109	0.104
NFL	1992	6307									
NFL	1993	7025									
NFL	1994	9936									
NFL	1995	11415									
NFL	1996	13971									
NFL	1997	8670									
	Ave	10862	3078	2869	8306	4121	180	15476	18554	2.202	0.456
	Min	3371	272	553	745	167	0	1581	2351	0.111	-2.199
	Max	21199	24990	7429	22416	15058	1785	34028	56551	7.898	2.067

Table 3.1.2-16: Run reconstruction for North Fork Lewis River naturally spawning fall chinook. The recruits include cumulative ocean harvest impacts.

Stock	BY	S_{adult}	R_2	R_3	R_4	R_5	R_6	R_{adult}	R_{total}	R_{obs}/S	Ln(R/S)
NFL	1964	16857	2092	3390	27120	6135	0	36645	38738	2.298	0.832
NFL	1965	7927	1088	1928	6977	1331	0	10236	11324	1.429	0.357
NFL	1966	11627	375	1947	8451	10272	0	20670	21045	1.810	0.593
NFL	1967	9711	1329	2423	39521	3614	0	45558	46887	4.828	1.574
NFL	1968	7160	25688	8403	30794	7952	0	47149	72837	10.173	2.320
NFL	1969	4986	883	2728	18437	3257	0	24422	25305	5.075	1.624
NFL	1970	4130	1126	3945	7789	6712	0	18446	19572	4.739	1.556
NFL	1971	19926	7474	3040	23149	1976	0	28164	35638	1.789	0.581
NFL	1972	18488	2556	4609	6025	3426	0	14060	16616	0.899	-0.107
NFL	1973	9120	1976	1449	11949	1906	0	15304	17280	1.895	0.639
NFL	1974	7549	2089	4002	6233	2453	0	12688	14776	1.957	0.672
NFL	1975	13859	2014	2541	11735	3791	0	18068	20082	1.449	0.371
NFL	1976	3371	1417	3019	23415	2486	23	28943	30360	9.006	2.198
NFL	1977	6930	2200	5666	19473	3610	0	28748	30948	4.466	1.496
NFL	1978	5363	279	2052	3978	4040	202	10272	10551	1.967	0.677
NFL	1979	8023	3067	3310	12261	4018	31	19620	22687	2.828	1.039
NFL	1980	16394	1749	1607	7329	2104	152	11192	12941	0.789	-0.237
NFL	1981	19297	2063	1917	6225	3349	36	11528	13591	0.704	-0.351
NFL	1982	8370	1880	2868	11175	6574	0	20617	22497	2.688	0.989
NFL	1983	13540	3496	6095	14352	9541	293	30281	33777	2.495	0.914
NFL	1984	7132	4204	6559	12383	18556	2200	39698	43902	6.156	1.817
NFL	1985	7491	5715	3773	9533	12133	932	26370	32086	4.283	1.455
NFL	1986	11983	3524	3313	7808	8981	599	20701	24225	2.022	0.704
NFL	1987	12935	2704	587	5109	3512	564	9772	12476	0.964	-0.036
NFL	1988	12059	1849	2019	4265	6193	516	12993	14842	1.231	0.208
NFL	1989	21199	781	642	815	210	41	1708	2489	0.117	-2.142
NFL	1990	17506	3283	2881	7375	11132	746	22134	25417	1.452	0.373
NFL	1991	9066	1256	1382	3535	5046		9963	11219	1.238	0.213
	Ave	11143	3148	3146	12400	5511	235	21284	24432	2.884	0.726
	Min	3371	279	587	815	210	0	1708	2489	0.117	-2.142
	Max	21199	25688	8403	39521	18556	2200	47149	72837	10.173	2.320

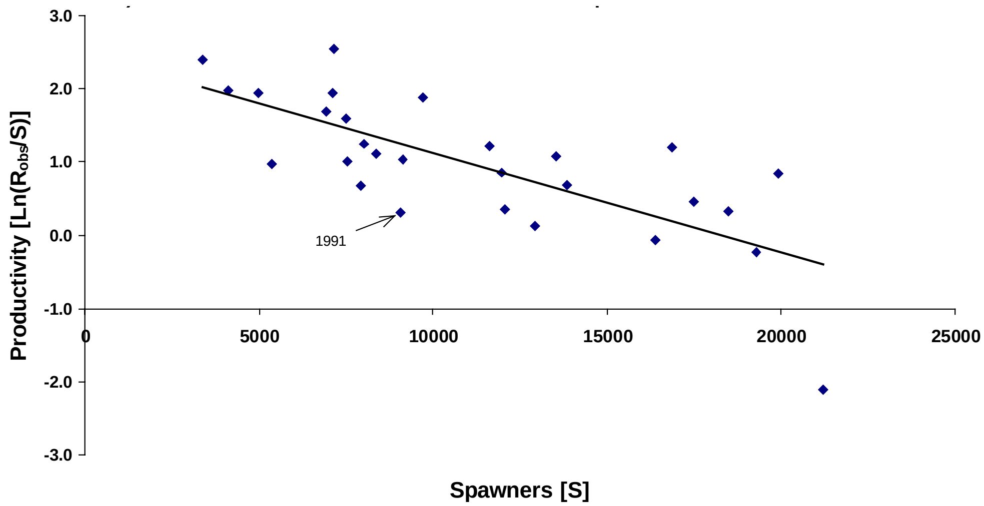

Figure 3.1.2-9: Productivity versus spawners for North Fork Lewis River fall chinook (BY 1964-91). Recruits include cumulative ocean harvest impacts.

Comparison Among Stocks

Spawner Trends

The HYURB stock exhibited an increasing trend in spawner abundance over the 1964-1991 period. The SRB stock exhibited a decreasing trend in spawner abundance, beginning in the late 1960s, to an extremely low level of spawners. The DES and NFL stocks did not exhibit noticeable increasing or decreasing trends in spawner abundance.

The SRB stock exhibited the largest coefficient of variation (CV; 133%) in spawner levels over the complete time series of information. For the other stocks the CV for spawner levels ranged from 35-63%.

Stock-Recruitment Relationships

For all stocks across all periods, the productivity value increased predictably when ocean impacts were included. The difference in productivity (with and without ocean impacts) was greatest for the NFL stock and least for the DES stock.

The productivity value was greater in the pre-1975 period than in the post 1974 period for HYURB and SRB. The productivity was less in the pre-1975 period than in the post-1974 period for the NFL stock. The contrast in spawner abundance was greater in the post-1974 period for NFL and HYURB fall chinook, which corresponded to a better fit to the stock-recruitment function in this period. The contrast in spawner abundance for the SRB stock was greater in the pre-1975 period and corresponded to a better fit to the stock-recruitment function (Table 3.1.2-10).

The HYURB and NFL productivity value, fit to all years, was intermediate of the productivity values fit to the other two periods. In contrast, the SRB productivity value fit to all years was less than the productivity values fit to the other two periods. The stock-recruitment fit to all years for SRB was the poorest for all stock and period combinations. The productivity values fit to all years is 2.5 or greater for the NFL, DES, and HYURB stocks with fits significant at the alpha 0.05. The productivity values for the stock recruit relationship fit to all years is 1.9 for SRB, and the fit is not significant.

For the SRB and NFL stocks, fit to the pre-1975 period, density dependence (beta) was significant at the alpha 0.05. In contrast, for this period HYURB stock density dependence was not significant (spawner escapement contrast was relatively small). For the HYURB and NFL stocks during the post-1974 period, density dependence (beta) was significant at the alpha 0.05. In contrast, for the post-1974 period SRB stock density dependence was not significant (spawner escapement contrast was relatively small). For the HYURB, DES and NFL stocks fit to all years, density dependence (beta) was significant at the alpha 0.05. For the SRB stock, fit to all years, density dependence (beta) was not significant at the alpha 0.05.

The spawners estimated to produce maximum recruitment (MSP), from the stock recruitment function, lie within in the range of observed spawners for all stocks and period combinations, except for the SRB stock fit to all years (Table 3.1.2-10). In this case, the estimate for MSP exceeds the maximum observed spawner level twofold.

Survival Rate Patterns

Trends in the survival rate indices (for Ricker fit to all brood years) (Figure 3.1.2-10) do not indicate any obvious level-shift in survival rate. The low brood year 1991 survival rates correspond to the largest observed deviation from the predicted recruitment function for the upriver SRB and HYURB stocks, but not for the downriver DES and NFL stocks (Figures 3.1.2-6 through 3.1.2-9). There were no discernable differences in the pattern of survival rate indices when ocean harvest impacts were included.

Conclusions

The NFL and HYURB stocks remained productive with a relatively good fit to the stock recruitment function over all brood years. In contrast, the SRB stock was less productive with a poorer fit to the stock recruitment function. The large uncertainty in the stock recruitment parameters for the SRB stock is partially due to the lack of contrast in spawner levels over a large portion of the time series. However, a stock recruitment function fit to the pre-1975 period for the SRB stock yielded a good fit and exhibited a productivity level comparable to the HYUB and NFL stocks. This large uncertainty in fitting the SRB stock recruitment function to all brood years greatly limits the applicability of the estimate of spawners needed to achieve MSP for management purposes. Although fit to fewer brood years, the fit to the stock recruitment function for DES stock yielded a good fit and also exhibited a productivity level comparable to the HYURB and NFL stocks.

Accounting for ocean harvest impacts increased the productivity of the stock recruitment relationship, but did not appreciably change the slopes. Further, there were no discernable differences in the pattern of survival rate indices when ocean harvest impacts were included in the recruitment estimates. There were no obvious level shifts in the time series of survival rate indices for any of these stocks. More detailed comparative analyses of temporal and spatial patterns in the productivity and survival rate indices of the four stocks should be completed (including contrasting these patterns with the extrinsic factors of hydro, hatchery, habitat, and harvest influences).

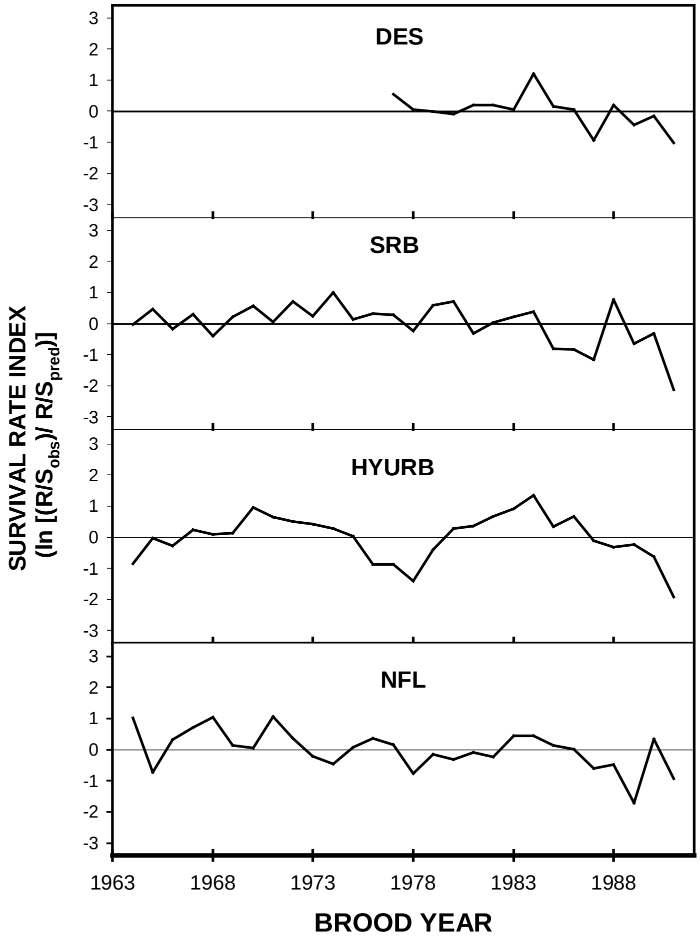

Figure 3.1.2-10: Survival rate indices by stock, BY 1964-91.

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3.2 Models

3.2.1 Passage Model Descriptions

CRiSP Model Description

Introduction and Overview

This section contains a brief description of the key components of the CRiSP model used for fall chinook modeling. In addition, calibration methods and results are presented along with the major assumptions used for model configuration.

Since data for model development and calibration are much sparser for fall chinook than for spring chinook, and since fall chinook migratory behavior is generally more complex than that of spring chinook, we chose to adopt a "top down" calibration approach for fall chinook modeling. That is, we used the NMFS project survival estimates to calibrate behavioral relationships instead of using these data for validation, as was done for CRiSP spring chinook. This approach ensures that the model output will reflect the most current (and only) survival estimates for Snake River fall chinook.

The strategy for this analysis was as follows. The fish behavioral parameters were calibrated to the 1995-1997 NMFS survival studies data. These behavioral parameters were then applied to our best estimates of historical dam and reservoir configurations along with historical temperatures, flows, and spills to produce the retrospective runs.

The NMFS survival studies provide two sets of estimates for the Snake River fall chinook: survival from release to Lower Granite Dam, and from Lower Granite Dam to Little Goose and Lower Monumental Dams. Since the fish were released as pre-smolts, the release to Lower Granite segment represents a rearing phase in addition to a migration phase. The segment from Lower Granite Dam to Lower Monumental Dam represents a migratory phase. Since these two segments are characterized by different fish behaviors, we calibrated the model separately for the two segments and produced separate results which were later combined to produce results from release through the hydrosystem. In addition, the upstream phase was further partitioned into a migration phase and a rearing phase. Model runs were produced that contained both rearing and migration phases and migration only.

Many assumptions were required to produce these runs. The CRiSP modeling team along with the FLUSH team agreed to an initial set of assumptions under the consultation of representatives from NMFS, ODFW, USGS and the Army Corps. In the future these assumptions will be analyzed by conducting sensitivity analyses based on alternative model configurations.

This report is divided into the following sections: *Major assumptions* will outline the assumptions and provide comments; *Calibration* will discuss the data used in the calibration and provide results, including parameter estimates. The results of the retrospective runs are presented in Section 3.3.1.

Major Assumptions and Conditions

As stated above, several assumptions were required in order to complete these runs. These assumptions are summarized below.

Bypass survival: At all dams except Bonneville, bypass survival was set to 0.88. This is based on paired release studies conducted by NMFS at Little Goose dam in 1997. The estimate is the mean of five replicates with substantial variability (Steven G. Smith, pers. comm.). Bypass mortality includes direct and indirect mortality but does not include mortality due to predation in the tailrace common to all passage routes, which is modeled separately. The bypass mortalities at Bonneville are based on Army Corps estimates. At Bonneville first powerhouse, bypass survival was modeled as 0.72; at the second powerhouse, bypass survival was modeled as 0.83.

Fish guidance efficiencies: FGE values are contained in Section 3.1.1 of this document.

Spill effectiveness: Due to time constraints, we used a simple model for spill effectiveness. At all dams except The Dalles, spill effectiveness was modeled as 1.0. At The Dalles, spill effectiveness was modeled as 2.0. Future runs will likely incorporate a more complex model based on more recent observations.

Direct transport mortality: Because little evidence exists for mortality of subyearling chinook on trucks and barges, direct transport mortality was set at 0.98, as was used for spring chinook modeling.

Predation mortality: We assumed that predation mortality has been reduced 12% since the early 1990's by the predator removal program. For CRiSP runs, this required reducing predators by 20% in Lower Granite Pool and by 27% in the lower reservoirs.

Modeling free flowing reservoirs in CRiSP: Several dams were not present in the early years of the retrospective runs. To model these years, we removed the dams and associated mortalities (including forebay and tailrace predation) and lowered the reservoirs to pre-impoundment levels. We then assumed the predator densities were unchanged in the reservoirs and applied the predation relationships derived for the full pool. Fish travel time was modeled utilizing the relationships obtained from full pool conditions but extrapolated to faster water travel times.

Release distributions: The release distribution at the head of Lower Granite Pool was based on the PIT tagging distribution of wild subyearling chinook in the free-flowing Snake. This assumes that the tagging effort results in a tagging distribution consistent with abundances of wild populations. For CRiSP the passage distribution at Lower Granite Dam was based on the average passage index for wild fall chinook for the years 1993-1997. Methods and plots were presented in a previous PATH document on the fall chinook web page.

Reservoir mortality. CRiSP assumes that the primary cause of reservoir mortality for fall chinook is predation. Thus once dam mortality is imposed, the remaining mortality is attributed to predation in the forebay, tailrace and main reservoir.

Another assumption of the CRiSP model is that behaviors observed in the upstream reaches can be extrapolated to downstream reaches. In other words, we have survival estimates only for release sites to Lower Granite and Lower Granite to Lower Monumental. These observed survivals are used to calibrate the predation submodel and the response to factors such as temperature. We then assume that the behavioral responses in the lower reservoirs are consistent with those in the upper reservoirs (Little Goose and Lower Monumental).

Calibration

Optimization routines were used to calibrate both migration rate and predation rate parameters. Both optimization routines operated by minimizing a merit function based on differences between observed travel time or survivals and ones generated by CRiSP under particular parameterizations. Also, since survival and travel time are not independent in CRiSP, the two components were calibrated iteratively until a common set of parameters led to best fits in both components.

Methods

<u>Migration Rate</u>

Migration rate parameters were estimated by procedures presented in Zabel et al. (1998). The procedure involves fitting increasingly complex migration rate models to the data and selecting the model with the level of complexity supported by the data. The simpler models have either a constant migration rate or a linear relationship with river velocity. The most complex model includes downstream acceleration and a seasonal flow term where fish use more of the river velocity for migration later in the season. The equation for the most complex model is

$$r_t = \beta_0 + \beta_1 \left[\frac{1}{1+\exp(-\alpha_1 (t - T_{RLS}))} \right] + \beta_{FLOW} \overline{V}_t \left[\frac{1}{1+\exp(-\alpha_2 (t - T_{SEASN}))} \right] \qquad \text{[Eqn. 3.2.1.1]}$$

The formulas inside of the square brackets are logistic equations that vary with time and attain maximal values. This form was selected to ensure that the equation does not produce results outside the range of observations. The α's are slope parameters, and the T's are inflection points. The first set of terms describes a downstream acceleration independent of flow. β_{MIN} (which is equal to $\beta_0 + \beta_1 /2$) is the minimum flow-independent migration rate, and β_{MAX} (which is equal to $\beta_0 + \beta_1$) is the maximum flow-independent migration rate. The second term describes the flow-dependent migration rate, with the flow dependency increasing as the season progresses. V_t is the water velocity, and β_{FLOW} is the maximum proportion of river velocity used for migration. See Zabel et al. (1998) for a fuller explanation of the model and parameters.

This equation involves the fitting of 6 migration rate parameters (β_{MIN}, β_{MAX}, β_{FLOW}, α_1, α_2, and T_{SEASN}). The simpler models are obtained by "turning off" some of the parameters. As mentioned above, we begin by fitting the simplest model and sequential increase complexity by adding parameters. If an added parameter does not substantially improve the model performance, we chose the simpler model. In addition, we estimate the rate of population spreading (Zabel and Anderson 1997) based on the spread of arrival distributions at observation sites.

Predation Mortality

As stated above, the "unexplained" mortality (i.e., mortality remaining after dam mortality and a negligible amount of nitrogen mortality) is assumed to be due to predation. Thus to calibrate CRiSP, the predation rate parameters were tuned until CRiSP-generated survivals best agreed with observed survivals.

The general form for predation rate is

$$r_t = \alpha \cdot P_t \cdot f(T) \qquad \text{[Eqn. 3.2.1.2]}$$

where

α is the predator activity coefficient that varies by reservoir zone,
P_t is the predator density, and
$f(T)$ is the temperature response equation.

For the temperature response equation, the sigmoidal form from Vigg and Burley (1991) was employed:

$$f(T) = a/(1+\exp(b-cT)) \qquad \text{[Eqn.3.2.1.3]}$$

We assumed that the temperature response function did not vary by location (tailrace, forebay, or mid-reservoir) but that the predator activity (which is related to consumption rate) did. Further, to simplify the fitting procedure, the predator activities were assumed to be proportional to those observed in the different reservoir zones for July and August (Rieman et al. 1991 and Anderson et al. 1996). In other words, we did not fit consumption rates separately for the forebay, tailrace or mid-reservoir (the resolution of the data would not support this) but assumed the relative consumption rates in these zones were the same as observed elsewhere. Thus, we fit the three parameters of Equation 3.2.1.3: a, which scaled the level of predation, and b and c, which together determined the temperature response. Note also that a and α are confounded, so they are fit together.

Predator abundances are based on the predator index studies performed by USFWS, ODFW, and WDFW from 1990-1993. The predator abundances for the major predators (northern pikeminnow, walleye, and smallmouth bass) are described in terms of northern pikeminnow equivalents. A table of predator abundances is contained in Anderson et al. (1996). For the prospective model runs, the predation mortality is assumed to decrease by 12% due to the predator removal program.

Travel Time Results

Plots of observed versus modeled median travel times are presented in Figure 3.2.1.1. The estimated migration rate parameters are contained in Table 3.2.1-1.

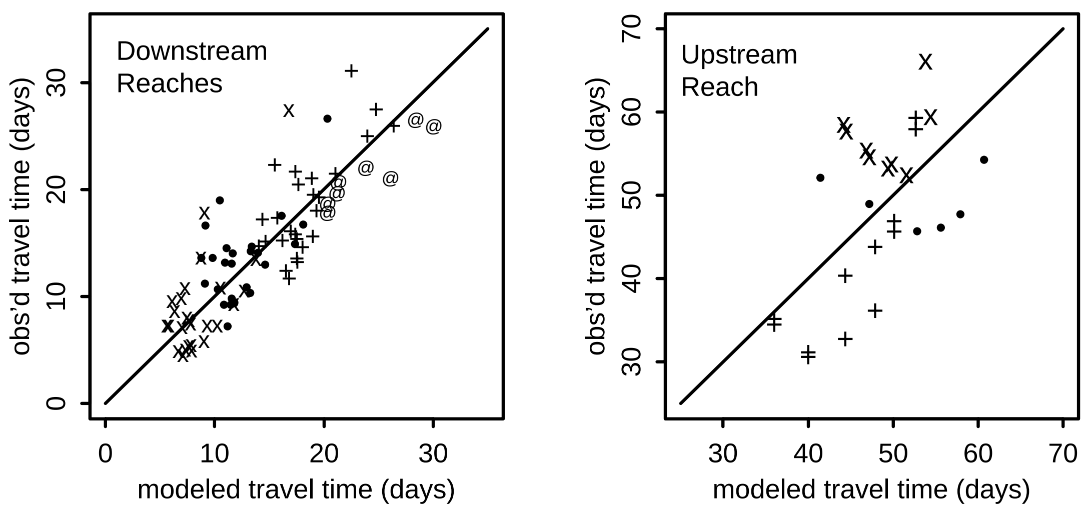

Figure 3.2.1-1: Observed versus modeled median travel times for upstream and downstream reaches. For the downstream reaches, x represents travel times from Lower Granite to Little Goose; * represents Lower Granite to Lower Monumental; + is Lower Granite to McNary; and @ is Lower Granite to Bonneville (1997 only). For the upstream reach, * represents 1995, x = 96, and + = 97. Observed travel times in the upstream reach are obtained from the NMFS survival study Snake River releases. For the downstream reaches, observed travel times were obtained by pooling all NMFS survival fish into weekly cohorts at Lower Granite.

For the upstream reach, more complex models were not supported by the data, and a linear flow model was selected. Clearly uncertainty exists in this flow/travel time relationship (Figure 3.2.1.1). For the downstream reaches (Lower Granite to Bonneville), a more complex model was supported by the data and provides a reasonable fit ($R^2 = 0.625$).

Table 3.2.1-1: Migration rate parameters (from Equation 3.2.1.1) for upstream and downstream segments. Units are miles and days. σ^2 is the rate of population spreading (Zabel and Anderson 1997) in units miles^2/day.

Segment	Parameter Estimates							Resid. SS	R^2
	β_{MIN}	β_{MIN}	α_1	β_{FLOW}	α_2	T_{SEASN}	σ^2		
rel-lgr	0.344			0.025			1.69	1735.97	.315
lgr-bon	0.277	31.96	0.047	0.790	0.179	198.2	115.2	898.50	.625

Survival Results

Plots of observed versus modeled survivals for upstream and downstream (Lower Granite to Lower Monumental) are provided in Figure 3.2.1.2.

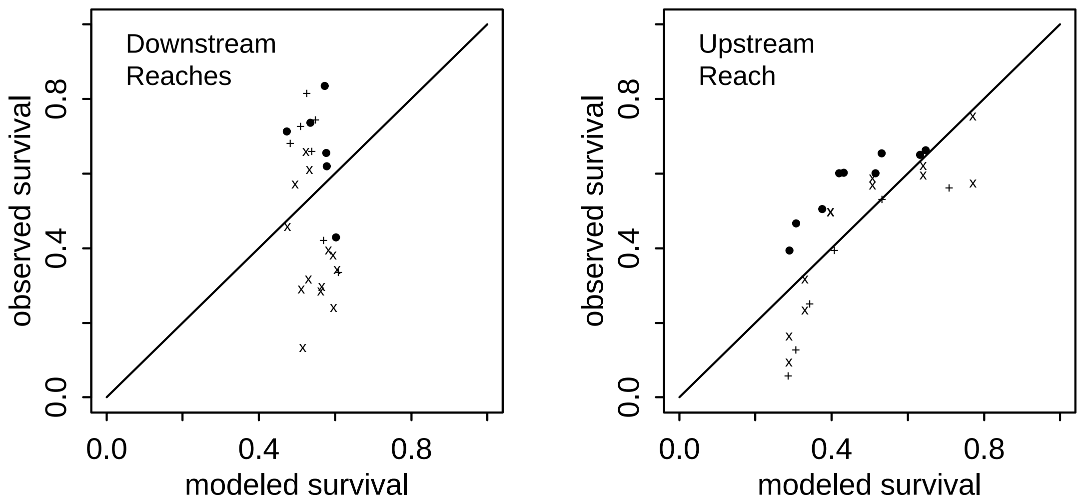

Figure 3.2.1-2: Observed versus modeled survivals for the upstream and downstream reaches. In both plots, * represents 1995, += 1996, and x=1997. The observed survivals were obtained from NMFS reports and from William Muir, 1997 data).

For the upstream reach, the fit of modeled survivals to the data was generally good except for the low observed survivals. The points with low survival that fell below the one-to-one line were late season releases. CRiSP predicted that survivals would begin to increase later in the season as temperatures decreased, but in reality survivals continued to decrease with time. Few wild fish initiated migration this late in the season, so this trend did not affect the retrospective runs.

For the downstream reach, variability existed in the survivals, but the model did not describe this variability. The model results were consistent with other analyses that did not detect trends in the survivals associated with environmental covariates (Muir and Smith 1998). The models cannot detect trends that don't exist. In this case, the model will produce results that are consistent with observed mean levels of survivals.
The estimated parameters are contained in Table 3.2.1-2. The parameters yield a maximum consumption rate at approximately 22^0C. Reservoir predator densities are approximately 400-600 predators/km^2. Applying the LGR-LMO reach parameter estimates from this table with these predator densities to equation 3.2.1.3 yields a daily survival rate of approximately 0.965 per day in the reservoir.

Table 3.2.1-2: Parameter estimates for the predation rate equations (Equations 3.2.1-2 and 3.2.1-3)

Reservoir Zone	Parameter Estimates		
	a*α (10^{-5})	*b*	*c*
reach: rel-lgr	7.17	14.8	0.80
reach: LGR-LMO	7.95	15.6	0.84
forebay: LGR-LMO	12.72		
tailrace: LGR-LMO	77.29		

Rearing and Migration Survival

As mentioned above, we produced two sets of model runs, one that included migration and rearing phases and one that included migration only. Since survival estimates from release to Lower Granite included both rearing and migration phases, it is necessary to estimate rearing mortality to be removed from total survival to produce the "migration only" model runs.

We decided to keep the approach to modeling rearing mortality as simple as possible since little data exists for calibration. The calculations required two assumptions: first, that the migration rates through Lower Granite Pool were the same as those through Little Goose Pool; second, that the mortality rate was equal in the rearing and migration phases. Based on the second assumption, the total survival from release to Lower Granite can be partitioned into rearing and migration survivals based on the proportion of time rearing and migrating.

The first step in the analysis was to compute the median travel time of fish migrating in Lower Granite Pool (t_M) based on the median migration rate of wild fall chinook migrating through Little Goose Pool for the years 1995-1997. This travel time was compared to the total median travel time from release to Lower Granite (t_T) to produce a proportion of time migrating of 0.15. The time spent rearing was also calculated as $t_R = t_T - t_M$.

The total survival, S_T, can also be partitioned as

$$S_T = S_M \cdot S_T \,, \qquad \text{[Eqn. 3.2.1.4]}$$

Where S_M is survival during migration, and S_T is survival during rearing. Based on the second assumption above, this equation can be expressed as

$$\exp(-rt_T) = \exp(-rt_M) \cdot \exp(-rt_R) \qquad \text{[Eqn. 3.2.1.5]}$$

Based on the proportion of time migrating mentioned above, survival during migration is calculated as

$$S_M = \exp(-0.15 \cdot r_T) = (S_T)^{0.15} \qquad \text{[Eqn. 3.2.1.6]}$$

Discussion

One issue concerning these model runs is that we used point estimates for our behavioral parameters. These point estimates do not incorporate the uncertainty in the data. As a result, relationships such as flow/travel time and predation rate/temperature may be overstated. This is particularly important because in some cases the observed relationships are weak. In the future, it will be beneficial to examine the effects of these derived relationships by conducting model runs with the relationships "turned off" or by performing formal sensitivity analyses on key parameters. Another approach to analyze this problem would be to utilize a Bayesian framework where a number of different relationships are allowed as dictated by the data.

Removing rearing mortality to produce the "migration only" runs requires several assumptions. At the current time, little hard data exists to support these assumptions. Also, the transition from the rearing phase to migration is likely gradual in fall chinook, with fish continuing to rear as they migrate through Lower Granite reservoir. The issue of what phases in the life history to include in the passage model runs needs to be further addressed. I would argue for a single set of model runs that require the fewest assumptions and can be best supported by available data.

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Fall FLUSH Model Description

Fall FLUSH

Fall FLUSH was developed to estimate the survivorship of sub-yearling chinook from the beginning of the Lower Granite (LGR) reservoir through to the Bonneville Dam tailrace. Fall FLUSH was developed as a mechanistic model to simulate changes in survivorship as a sub-yearling migrates through the hydrosystem. In Fall FLUSH, mortality arises from predation and dam passage. Dam passage routes include spill, bypass, turbines, and collection for transportation by truck or barge.

The model begins with an initial emigration start date represented as a normal distribution of daily cohorts at the head of LGR. The amount of time spent in a reservoir is determined by the applicable fish travel time (FTT) relationships and water velocities experienced. During the time required to travel through the reservoir, juvenile chinook are subject to predation. Once fish reach a dam they are forced to one of three different routes of passage; they are either spilled, bypassed or go through the turbines. After passing through the dam, sub-yearlings are then subject to predation in the first km immediately below the dam (defined as the boat-restricted zone or BRZ). The sub-yearling migration is modeled in this fashion from the head of LGR pool through eight hydroprojects ending at the Bonneville BRZ. The following finite difference equation summarizes losses to a daily cohort by each of these factors:

$$N_{i,t+1} = N_{it} - N_{it}(a_j S_{ijt} + b_j (PTrans_{ijt} + (1 - PTrans_{ijt} B_{ijt})) + c_j T_{ijt} + P_{ijt}) \qquad \text{[Eqn. 3.2.1-7]}$$

where

N_{it} = number of sub-yearlings in cohort *i* on day *t*
$a_{i,j}$ = probability of being spilled over dam *j* for cohort *i*
S_{jjt} = probability of mortality after being spilled over dam *j* on day *t* for cohort *i*
$b_{i,j}$ = probability of being bypassed around dam *j* for cohort *i*
B_{ijt} = probability of mortality after being bypassed at dam *j* on day *t* for cohort *i*

$c_{i,j}$ = probability of passing dam *j* through the turbines for cohort *i*
$PTrans_{ijt}$ = probability of cohort *i* being transported at dam *j* on day *t*
T_{ijt} = probability of mortality after passing through the turbines at dam *j* on day *t* for cohort *i*
P_{ijt} = probability of mortality due to predation in section *j* on day *t* for cohort *i*

Details on model development for each of these factors are described below, and a schematic is depicted in Figure 3.2.1-3.

Initial Emigration Distribution

The emigration distribution represents the date and proportion of the sub-yearling population (a daily cohort) begins their migration from the head of LGR reservoir. This distribution was determined from the number of PIT-tagged wild sub-yearling chinook released on each sample date from studies conducted from 1991-1997 (Connor et al. 1993-1998). We assumed the sampling distribution to be representative of the general migration distribution (W. Connor, pers. comm.). Several smaller sub-yearlings are recaptured in the area originally collected. After a sub-yearling reaches 85 mm it is rarely recaptured. Therefore, fish smaller than 85 mm are suggested to be rearing and not actively migrating. We used only the released dates of PIT-tagged fish greater than or equal to 85 mm to estimate the initial emigration distribution.

The initial distribution exhibited inter-annual variation of the mean emigration date. This variation is thought to be a result of differences in rearing environments with temperature largely influencing growth during rearing. This is supported by the strong relationship between the mean emigration date and the mean May 15-June 15 LGR dam temperature with:

$$ED = 195.19\left(1 - \frac{1}{20.11 - T}\right) \qquad r^2=0.85 \qquad n=6 \qquad \text{[Eqn. 3.2.1-8]}$$

where

ED = the trimmed mean (excluding the upper and lower 10% of the distribution) of the emigration distribution
T = the mean temperature from May 15-June 15.

The trimmed mean was used because of skewed distributions. A median could be used for the same purpose which produced a similar model with:

$$EstimatedMedian = 191.4\left(1 - \frac{1}{21.0 - T}\right) \qquad r^2=0.825 \qquad n=6 \qquad \text{[Eqn. 3.2.1-9]}$$

The relationship based on the trimmed mean was used to adjust the initial emigration distribution for a given water year. The mean standard deviation among years was used to generate a 61 day-wide normal distribution. In the model, the mean of this initial emigration distribution for a given year was determined from the temperature-adjusted mean.

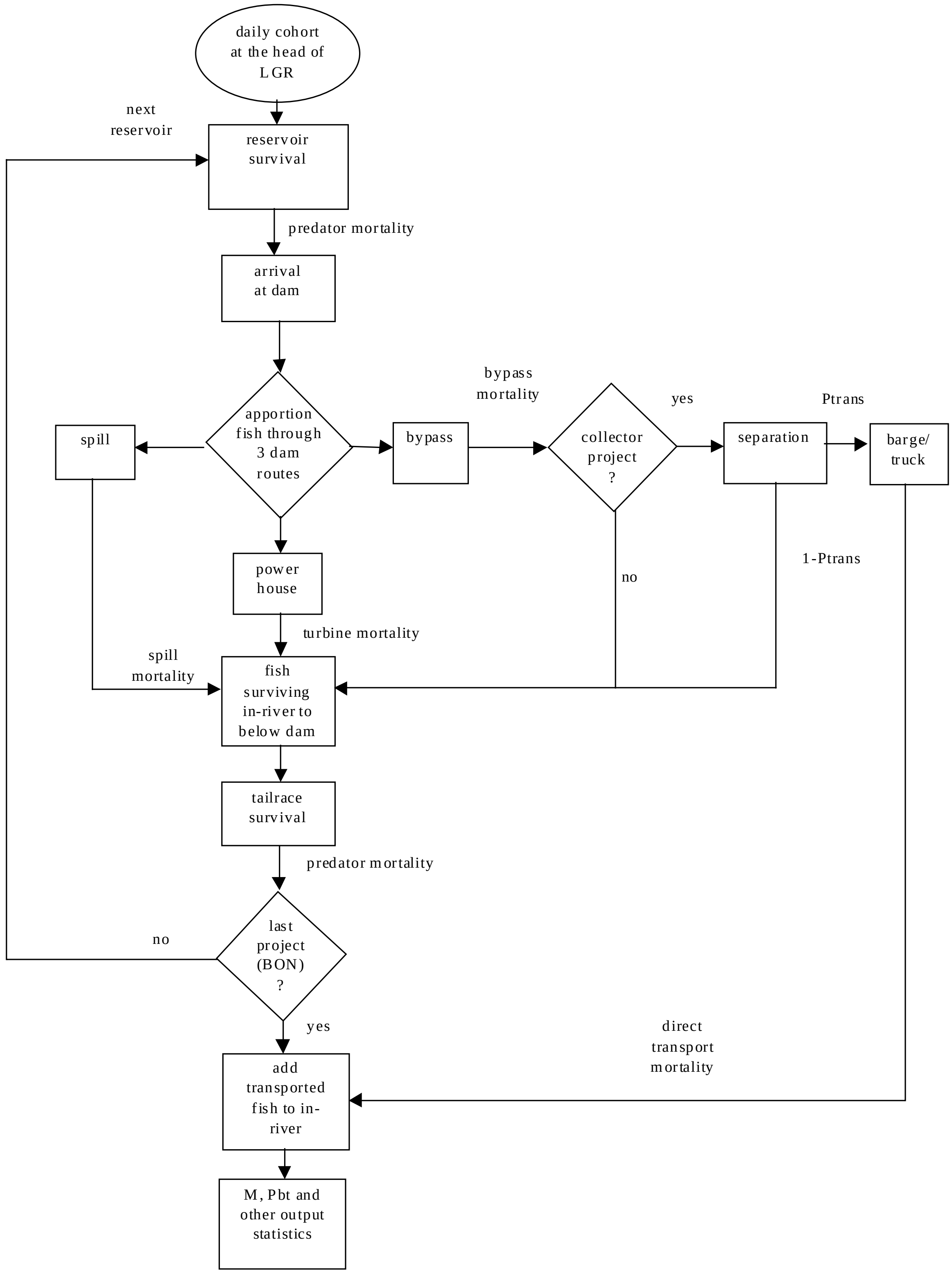

Figure 3.2.1-3: Schematic diagram of Fall FLUSH

Fish Travel Time Relationships

Hydrosystem sub-yearling chinook experience is defined temporally by the FTT relationships. The time required for sub-yearlings to migrate through a reservoir has been empirically estimated through PIT tag studies conducted from 1991-1997 by (Connor et al. 1993, 1994a, 1994b, 1996, 1997, 1998). Fish were tagged at several sites upstream from LGR. Tagged fish were then detected at downstream dams with date of detection recorded at each dam. The difference between release date and detection date was the time required to travel between these points. These estimates were regressed against temperature, discharge, water travel time (WTT), length at release, and date released to determine the best predictor(s) of FTT.

Once a cohort reaches the first reservoir, FTT decreases. The slow FTT in the first reservoir likely represents a behavior where fish are both rearing and migrating and reflects the impact of dramatically reduced water velocity relative to the free-flowing sections. Thus, observed FTT in LGR reservoir is much slower than in the subsequent reservoirs. As LGR reservoir is the first reservoir encountered after 1975 a relationship specific to this reservoir was applied to FTT. Specific FTT relationships were developed for the LGR dam to MCN dam reach and the MCN dam to BON dam reach. The LGR relationship was modified in years before the LMO LGO, and LGR dams were in place as fish likely had different migrating behaviors in the free-flowing section of stream now inundated by reservoirs. The first reservoir encountered prior to 1970 was IHR. We assumed that FTT in IHR was slower prior to impoundment of the upper three dams than in the post-impoundment period. We therefore applied the LGR reservoir FTT relationship to IHR reservoir. In the free-flowing section, we assumed FTT was equal to WTT, and survival was either fixed at 0.9997/mile or modeled during this FTT. The same approach was used when LMO and LGO came on-line. A similar approach was used prior to the construction of JDA where FTT was equal to WTT in the JDA reach.

LGR Reservoir FTT

As sub-yearling chinook spend time rearing while migrating, travel times vary greatly with smaller fish generally having slower migration rates before arriving at LGR dam. To account for the time spent rearing, we assumed arrival at the beginning of LGR reservoir occurred when fish reached 85 mm (Connor et al. 1993). WTT was the best single predictor of FTT within a year (Table 3.2.1-3). The coefficient of variation between years for the intercept parameter was much greater than the coefficient of variation of the slope parameter suggesting that WTT explained the relative differences in FTT but some other factor determined the absolute difference in FTT between years. The inter-annual differences in FTT were assumed to be a response to differences in inter-annual seasonal temperatures. An ANCOVA suggested that 2 different classes of average yearly temperature (averaged over May 15-June 15) separated the effects of WTT on FTT (significant intercept parameters and homogenous slope parameters) (Figure 3.2.1-4). The final FTT relationships in LGR reservoir was:

$FTT = 11.19WTT - 1.46$ in years where average temperatures were less than 14 °C

and

$FTT = 11.19WTT - 39.44$ in years where average temperatures was greater than 14 °C with r^2=0.524 and p<0.0001 n=315.

Table 3.2.1-3: Fish Travel Time (FTT) as function of Water Travel Time (WTT) and temperature. WTT and temperatures are the averaged over the migration time of release group of PIT-tagged wild subyearling chinook greater than 85mm

FTT vs WTT no rearing (only >85mm)				
	N	**intercept**	**parameter**	**adjrsq**
91				
92	15	-12.478	4.936	0.226
93	39	-10.277	9.959	0.524
94	53	-38.321	10.98	0.452
95	120	-6.86	13.743	0.615
96	40	1.689	9.098	0.439
97	42	-7.45	13.738	0.613
lnFTT vs TEMP (C) no rearing (only >85mm)				
	N	**intercept**	**parameter**	**adjrsq**
91				
92	15	-4.188	0.424	0.06
93	39	-2.578	0.337	0.352
94	53	-0.968	0.24	0.287
95	120	0.213	0.183	0.387
96	40	1.733	0.096	0.229
97	42	-0.897	0.229	0.245

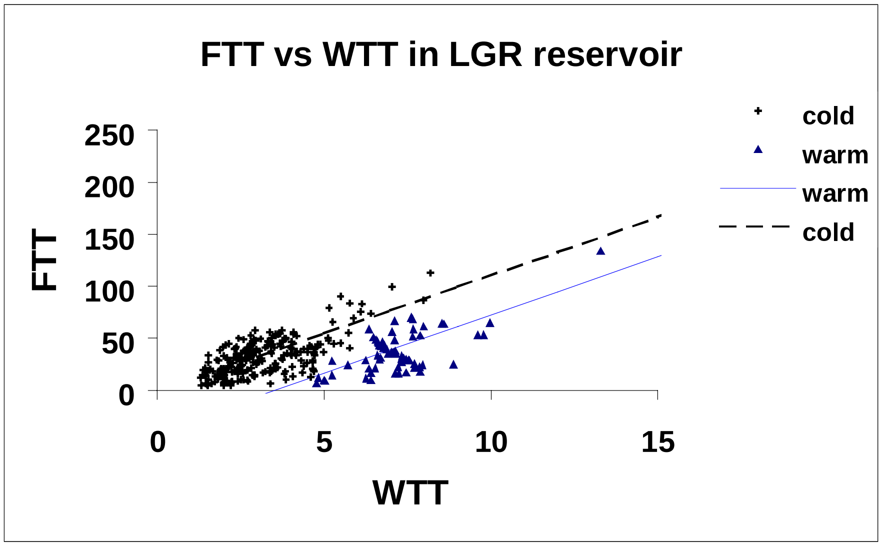

Figure 3.2.1-4: Fish travel time (FTT) for subyearling chinook as a function of water travel time (WTT) for cold years mean May 15-June 15 LGR temperature was less than 14°C) represented by the open circles and dashed line and warm years (mean May 15-June 15 LGR temperature was greater than 14°C) represented by the filled triangles and solid line.

LGR Dam to McNary Dam

FTT was determined from LGR to McNary as the difference in detection dates of PIT-tagged fish from these two projects. Wild subyearling chinook FTT was regressed against WTT producing a FTT from LGR to McNary modeled as:

$$FTT = 5.121e^{0.0855WTT} \qquad r^2=0.275 \;\; p<0.0001 \;\; n=78 \qquad \text{[Eqn. 3.2.1-9]}$$

McNary Dam to Bonneville Dam

FTT was also determined from MCN to BON in 1997 for Snake River subyearling chinook, however, due to the low sample size of individual wild fish detections, hatchery and wild fish were used in the FTT relationship for this reach. FTT for this reach was described by WTT with:

$$FTT = 1.434e^{0.1778WTT} \qquad r^2=0.33 \;\; p<0.0001 \;\; n=181 \qquad \text{[Eqn. 3.2.1-10]}$$

Predation

The FTT relationship defines the amount of time a sub-yearling remains in a reservoir. Throughout the duration of time spent in the reservoir, sub-yearlings are subject to a host of predators. Every day spent in the reservoir, a portion of a cohort is lost to these predators. After the fish travel through the reservoir they encounter a dam. After passing the dam they are subject to predation in the BRZ.

Predators in the BRZ are modeled with different consumption rates than in the reservoir as predators have been documented to alter foraging behaviors in this area. Predation rates are generally higher in the BRZ as predators exhibit different numeric and functional responses. The high concentration of stressed and disoriented juvenile salmonids in the BRZ after passing the dams either through bypass, spill, or turbines is thought to be responsible for the increased consumption rates. Predation in the BRZ in Fall FLUSH occurs in 1 day and is thus independent of FTT. All mortality in the BRZs and reservoirs is due to predation. The percent mortality due to predation in the migration corridor was modeled as:

$$P_j = b_z C\max{}_{jt}\, \alpha_z Q_z \delta_k AI_{kj} \qquad \text{[Eqn. 3.2.1-11]}$$

where

P_j = percent mortality in reservoir or BRZ *j*
b_z = parameter where *z* is the zone (MCN BRZ or JDA reservoir)
$C\max_{jt}$ = the maximum potential consumption rate as a function of temperature at dam or reservoir *j* at time *t*
α_z = the percent active predators for *z*
Q_z = the squawfish population estimate for section *z*
δ_k = the conversion from squawfish consumption to species *k* consumption
AI_{kj} = the abundance index for species *k* in section *j*

Details on parameter estimations are found in Bouwes (1998). Abundance indices for northern squawfish, walleye, and smallmouth bass, prior to the predator removal program for all BRZs and reservoirs, were used to determine relative predator population abundance in the model. The consumption conversion index describes the relative rate of consumption of smallmouth, walleye, or catfish to squawfish.

Predator Population Estimates

Between 1983-1986 an intensive predator assessment program was implemented in JDA reservoir (Poe and Rieman 1988). Mark-recapture population estimates were made for northern squawfish, smallmouth bass, and walleye (Rieman et al. 1991). Predator abundance and consumption rate estimates have been monitored at the majority of the hydroprojects' reservoirs and BRZs from 1991 to the present (Zimmerman and Parker 1995; Ward et al. 1995; Ward 1997). Predator population estimates were indexed to Catch Per Unit Effort (CPUE) information relative to the mark-recapture estimates in John Day reservoir.

Catfish diet information suggests that salmonids often represent a large proportion of their diet (Vigg et al. 1991). Estimates of catfish populations have not been formally determined through mark-recapture estimates, and thus the impact of catfish on salmonid mortality has been largely ignored (Poe et al. 1994). Gill net information, however, suggests that catfish are highly abundant from LGR reservoir to MCN (Zimmerman and Parker 1995). We compared bottom and surface gillnet-pooled CPUE information for northern squawfish and catfish (Zimmerman unpublished data) in the Snake River. Catfish CPUE was approximately 5 times that of squawfish in these areas. This assumes equal vulnerability for both species to gillnets. As a conservative estimate, we assumed that catfish abundance was twice squawfish abundance in the Snake River. We also conservatively assumed catfish were not present downstream of MCN dam.

Consumption Estimates

Consumption rates for the major predators in the MCN BRZ and in JDA reservoir from 1983-1986 were estimated through a gut evacuation technique developed by Swenson and Smith (1973) and reported in Poe et al. (1991), Vigg et al. (1991), and Reiman et al. (1991) (see Poe and Rieman 1988). These observed northern squawfish consumption rates were compared to salmonid density to determine whether predation foraging patterns adhere more closely to Type l or Type III functional response models adjusted for temperature (Bouwes et al. in prep.). As a Type 1 (linear) response of predators to prey density provided a good fit to the observed consumption estimates (Bouwes et al. in prep.), we assumed that predation rates were independent of prey numbers (Juliano 1993).

Water temperature constrains physiological processes of fish. The rate at which a fish can consume prey is in part dependent on temperature. These physiological limits have been described in depth by bioenergetics models (Ney 1990, Hewett and Johnson 1992). In Fall FLUSH, *C*max was used to adjust for the effects of temperature on consumption rates of squawfish. *C*max is the maximum potential consumption rate of a predator at a given temperature. A gamma function described the relationship between *C*max and temperature as determined in the laboratory by Vigg and Burley (1991). This variable was used to adjust observed consumption estimates when regressed against prey density. The following equation related consumption rates of squawfish to prey density for JDA pool and MCN BRZ:

$$consumption = 0.442 * C\max * preydensity$$ in JDA reservoir with r^2=0.55 and $p<0.0001$

and

$$consumption = 0.559 * C\max * preydensity$$ in MCN BRZ with r^2=0.65 and $p<0.0001$

These equations were used to predict consumption rates in reservoirs or BRZs for all projects for northern squawfish. Consumption rates of other species were represented as a proportion of squawfish consumption. Rieman et al. (1991) observed smallmouth bass consumption rates at an average of 0.28 that of squawfish consumption rates. Walleye consumption rates were set equal to 1.415 times that of squawfish (Rieman et al. 1991). Catfish consumption rates were 0.71 that of squawfish (Vigg et al. 1991).

Direct Dam Mortality

FTT for each cohort is tracked in FLUSH. Direct dam mortality is 0 while the fish migrate through the reservoir. On the day when fish encounter a dam, mortality is apportioned to each route of dam passage. Fish can pass dams through 3 different routes: spill (*S*), bypass (*B*), and turbines (*T*). The probability of reaching each route is represented by *a, b, c*, respectively where:

$$a = f_s / f * SS$$
$$b = (1 - a) * (FGE)$$
$$c = (1 - a) * (1 - FGE)$$

with

f	=	water flow rate past the dam (including spill)
f_s	=	water flow rate over spillway
SS	=	spill effectiveness
FGE	=	fish guidance efficiency (proportion of smolt diverted into collection facility-not including spilled smolt)

FGE values vary by project and year.

Once a cohort reaches a dam the probability of mortality for each route depends on the different passage assumptions. The model assumes passage mortality for S = 0.02 and T = 0.1. Paired releases above and below the bypass system suggest that mortality from the bypass facility is 12.0% (Muir 1998). This estimate, however, includes losses due to predation in the BRZ. To avoid applying mortality twice to the same fish we subtracted predation mortality from the bypass mortality. A predation mortality of 4% was subtracted from the total mortality to produce a B=0.08.

Transportation

A fish entering the bypass system is either sent to a collection facility or back into the river below the dam. All fish entering the collection facility are subject to bypass mortality. A large proportion of the fish captured in the collection facility is transported. The probability of being transported (*D*) is zero if the cohort has not arrived at the dam or if transportation for the year has ended, otherwise *D* is the seasonal average for each project reported by the Army Corp of Engineers. The transported fish are subtracted from the cohort for the remainder of the migration through the passage corridor. Mortality due to transportation is assumed to be 2% based on anecdotal evidence and visual observations. Surviving transported fish are then added to the total in-river fish after the Bonneville tailrace to determine instantaneous direct mortality.

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3.2.2 Life Cycle Model Description

The life cycle model for Snake River fall chinook is explained in the next sub-section first in general terms, and then mathematically. This is followed by a sub-section describing additional model aspects needed for prospective simulation. We refer to the entire modeling apparatus by the acronym "fall BSM" (fall chinook Bayesian Simulation Model). Section 3.3.2 provides a summary of the results of exploring different variations in the structure of the life cycle model, and testing them against historical estimates of recruitment.

Snake River Fall Chinook Life-Cycle Model for Retrospective Analysis

General Description

The general forms of both the spring/summer and fall chinook life cycle models are similar, that is:

$$\text{Recruits} = \begin{bmatrix}\text{Stock Recruitment}\\ \text{Function}\end{bmatrix}\begin{bmatrix}\text{System}\\ \text{Survival}\end{bmatrix}\begin{bmatrix}\text{Post} - \text{Bonneville Survival}\\ \text{of Non} - \text{Trans. Fish}\end{bmatrix}\begin{bmatrix}\text{Climate}\\ \text{Factor}\end{bmatrix} \qquad \text{[Eqn. 3.2.2-1]}$$

Each of the terms on the right side of the equation are described below, focusing on fall chinook. The *stock recruitment function* considers the stock's productivity (Ricker *a*), the spawning level generating maximum recruitment (*1/b*), the current number of spawners (*S*), and a parameter to potentially account for less recruitment at low spawning levels (*p*). In the Snake River, we model only one fall chinook stock, as compared to seven spring/summer stocks.

System survival is the number of in-river equivalent smolts below Bonneville Dam divided by the population at the head of the first reservoir. The term "in-river equivalent smolts" adjusts for post-Bonneville mortality of **transported** fish. Survival from the first reservoir to Bonneville is estimated from the passage models (fall FLUSH or fall CRiSP, see Section 3.2.1). The passage models also estimate the proportion of smolts below Bonneville which were transported in each year.

The post-Bonneville survival of transported fish is computed using the parameter "*D*", which is the ratio of post-Bonneville survival of transported smolts to post-Bonneville survival of non-transported smolts.

In the spring/summer models, the *D* parameter was estimated using transport-control studies, and passage model estimates of the in-river survival of control fish. This provided several estimates of *D*, for all the years with transport experiments. With fall chinook, however, there are no transport-control studies from Lower Granite Dam. We therefore needed a different method to estimate *D*. The method chosen for prospective modeling was to either use the most likely value for *D* given the spawner recruit data, or to adjust this estimated value to correct for possible biases in the method used. In the fall chinook model, the same *D* value is used for every year, because it isn't possible to estimate specific *D* values for different years.

Post-Bonneville survival of non-transported fish considers a step-decline in survival either after brood year (BY) 1970[19] (presumed to be related to the start up of the lower Snake River dams), or after BY 1976 (presumed to be related to changes in ocean conditions, or the full operation of the Snake River dams). The magnitude of this step decline (parameter "*STEP*") is also estimated from the spawner-recruit data. How much of a step decline in survival is assigned by the model depends not only on the assumed year in which the decline begins, but also on which passage model is used to estimate system survival. As can be seen from Equation 3.2.2-1, if system survival is higher, post-Bonneville survival of non-transported fish will need to be lower to generate the same recruitment).

Which passage model is used also affects the ability of Fall-BSM to estimate the "*D*" and "*STEP*" parameter. According to the CRiSP model, a large fraction of Snake River fall chinook began to be barged in 1977 (brood year 1976), as shown in Figure 3.3.1-3. The FLUSH model assumes that most of the fall chinook smolts migrated after the end of the barging period in the late 1970s, and substantial transportation of these fish only began in 1980. With the CRiSP model, therefore, the onset of transportation in 1977 completely coincides with potential climatic changes in 1977 and full operation of the Snake River dams. Any of these factors could have caused an increase in post-Bonneville mortality. In mathematical terms, either a low "*D*" (poor post-Bonneville survival of transported fish) or a significant "*STEP*" (decreased survival of non-transported fish) could explain post-1976 declines in recruitment. The spawner/recruit data are therefore sometimes insufficient to provide precise estimates of **both** "*D*" and "*STEP*". Therefore, when simulating extra mortality hypotheses that have a BY 1976 *STEP*, the user needs to specify a value for *D* and let Fall-BSM model estimate *STEP*. An alternative hypothesis is to assume *STEP*=0.0, and let the model estimate *D*.

The *climate factor* accounts for changes in survival other than those due to the stock recruitment function, system survival, and post-Bonneville survival of non-transported fish. The approach which gave the best fit to historical data was to assume that year-to-year variations in Snake River fall chinook recruitment track changes in the Deschutes fall chinook stock. The Deschutes stock is considered by NMFS to be in the same *Evolutionarily Significant Unit* as the Snake River stock, which lends support to this approach (Meyers et al. 1998). Other approaches to estimating a climate factor were much less effective. These included using temperatures from five Canadian weather stations, indices of year-to-year variations from the spring/summer chinook analysis, and year-to-year variations in the Hanford fall chinook stock.

Mathematical Description

Mathematically, the fall chinook spawner-recruit model is a hybrid of the alpha/delta models used for spring/summer chinook (see *PATH Weight of Evidence* report, Section 4.2.2). It is a Ricker model generalized to permit depensatory mortality, year-effects, in-river passage mortality, and post-Bonneville "extra mortality." The equation is:

$$ln(R_t) = a - bS_t + (1+p)ln(S_t) - M_t + yr\text{-}effect - STEP + ln(DP_t + 1 - Pb_t) + e_t \quad \text{[Eqn. 3.2.2-2]}$$

[19] The 1970 brood year corresponds to the 1971 migration year for fall chinook.

for which:

- a = Ricker "a" parameter (estimated from spawner/recruit data),
- b = Ricker "b" parameter (estimated),
- p = depensation parameter, to potentially reflect changes in survival at low spawner abundance (estimated),
- *STEP* = post-Bonneville mortality of non-transported smolts, which is a step function with values *STEP=0.0* for brood years prior to either 1976 or 1970 (depending on model choice) and takes on a constant value for subsequent brood years (either specified or estimated),
- D = ratio of post-Bonneville survival of transported smolts to post-Bonneville survival of non-transported smolts, which is assumed to be a constant during all years of transportation (either specified or estimated),
- *Pb_t* = proportion of smolts below Bonneville which were transported in year "t" (provided by a passage model),
- *M_t* = total direct in-river and transport mortality to below Bonneville in year "t", which may include mortality in rearing areas, prior to migration in river (provided by a passage model),
- *yr-effect* = year-effect accounting for mortality sources other than those specified by other parameters in the model (see below),
- *R_t* = number of returns to the Columbia River in the absence of ocean harvest data),
- *S_t* = number of age 3+ spawners in year "t" (data), and
- *e_t* = normally distributed random variable, with mean of zero.

Total in-river and post-Bonneville mortality is given by the sum of terms, *[–M_t –STEP + ln(DPb_t + 1-Pb_t)]*. *STEP* is always zero prior to 1970, and the *ln(Dp_t + 1-Pb_t*) term dissolves to zero in years without transportation (i.e., when *Pb_t* = 0). When exponentiated, , *[–M_t + ln(DPb_t+1-Pb_t)]* equals the system survival in year *t*, including post-Bonneville survival of transported fish. The derivation is given in Section 3 of Appendix A of the *Preliminary Decision Analysis* Report for spring/summer chinook.

Potentially, the parameters to estimate include *a, b, p, yr-effect, STEP, and D*. It is assumed that *S_t, Pb_t, and M_t* are input data measured without error. Both measurement error in the returns, *R_t*, and process error (natural variation) contribute to the error term *e_t*. Measurement error can affect the accuracy of other estimated parameters.

We explored a number of ways to estimate year effects (see Section 3.3). Based on the retrospective analysis described in Section 3.3, we chose to represent the year-effect for Snake fall chinook as a factor proportional to residuals from a Ricker model fitted to the Deschutes fall chinook stock. This factor could have been either negative or positive, but the data showed a positive correlation between the two stock groups. That is, if no other factors changed from year to year, years with better than expected recruitment in the Deschutes stock will also have better than expected recruitment in the Snake River stock. In probability notation, this means we need to model the joint posterior density of both the Deschutes and Snake River models (i.e., how the recruitment of these two stocks covaries). Formally the random variable:

$$e\_t = g * e\_t(deschutes) + e'\_t \qquad \text{[Eqn. 3.2.2-3]}$$

for which we assume that the *e_t(deschutes)* and *e'_t* are independent normal random variables. With this assumption about the random variables, we identify the "yr-effect" as a component of the *e_t*. The parameter g (the proportionality factor mentioned above) is an another parameter to be estimated. With the structure shown in Equation [3.2.2-3]. the "yr-effect" term in Equation [3.2.2-2] drops out, and is absorbed into the *e_t* term. We wrote Equation [3.2.2-2] as is to reflect other possible approaches to estimating the year effect (described below).

The values of all parameters are selected so as to maximize the likelihood of zero residual variation in both stocks (i.e., maximize the amount of variation explained by the two stock recruitment models, and minimize the unexplained noise). Maximizing the probabilities of the residuals *e_t* and *e'*_t is equivalent to minimizing the unexplained noise, because the probability distributions for *e_t* and *e'_t* are normal distributions with a mean of zero.

The posterior density (or probability distribution to be maximized) is basically a generalization of the likelihood function, which we can write as:

*Pr({e_t(deschutes)})*Pr({e'_t})* Pr(priors)* [Eqn. 3.2.2-4]

In application this involves simultaneous fitting of Ricker-type models to the Deschutes and Snake River stocks. The Deschutes region fall chinook and Hanford region fall chinook are modeled with simple Ricker-type models.

The prior distributions for all parameters are taken to be uniform distributions (i.e., the likelihood of any particular parameter value is determined only by the data), except for the Ricker "b" coefficient for the Snake River stock. For this "*b*" parameter, we have independent historical information which can be used to constrain the values assigned by the fitting procedure. At the PATH April 1998 workshop, we decided to assume a prior on that "*b*", which is implemented in the model as:

ln_b ~ N(ln(1/5000),.25) [Eqn. 3.2.2-5]

That is, other information suggests that recruitment is maximized at 5000 spawners (Schaller and Conney 1992). Estimates of the maximum recruitment level range from 4800 (Schaller and Cooney) 1992) to 7140 (Connor 1994). However, the BSM fits to spawner/recruit data are not very sensitive to this assumption (e.g., using a maximum recruitment prior of 10,000 spawners generated very similar results).

The Hanford stock was included in the current life-cycle modeling, not because it provides retrospective information about the Snake River stock, but because prospective in-river harvest rules depend on the Hanford stock performance. To date, we have not used the Lewis River spawner/recruit data in Fall-BSM. The Deschutes stock was included because it was effective as an estimate of year effects when fitting the Snake River stock (Eqn. [3.2.2-4]). Other candidate factors for the year effect parameter included Spring/Summer chinook MLE estimates of year-effect, PAPA index of changes in ocean conditions, and average ocean temperature data for five Canadian stations, but these resulted in significantly poorer fits to the spawner/recruit data.

Because the spawner/recruit data cannot estimate both "*D*" and "*STEP*", we used two versions of the Fall-BSM model. In one version of the model (called "Fall-S") "*D*" is set to a fixed input value, provided by the model user. The "Fall-S" version of the model assumes there is sufficient other information to specify a value for "*D*". It lets the spawner/recruit data determine the most likely value for "*STEP*" (for an assumed step-increase in extra mortality in 1976 – see discussion later on why STEP=0 if 1976 is assumed). In the other version of the model ("Fall-D"), "*D*" is estimated and "*STEP*" is set to 0. The "Fall-D" version of the model assumes that no step-change in extra mortality occurred in either 1970 or 1976, and uses the spawner/recruit data to determine the most likely value for "*D*".

Prospective Simulation Modeling

For prospective forecasts, 100-year simulations were made for each of the 4,000 samples taken of the posterior density of the parameters (the joint statistical distributions of all estimated parameters). Simulation year 1 is 1997, the first year spawning levels are forecasted. Each simulation begins with given inputs of spawning levels through 1996 and recruitment is forecasted beginning with brood year 1992.

Downstream Survival and Extra/Delayed Mortality

Parameters representing this source of mortality (*M_t, Pb_t, D, STEP*) are projected each simulation year given certain assumptions about how the river system is operated in the future (e.g., transportation, spill, drawdown, etc). During the forward simulation the parameters *M_t* and *Pb_t* are drawn from water years like those which occurred in 1976-1993 (brood years 1975-1992) based on selection relative to historical probabilities of years with similar unregulated water transit times.

Table 3.2.2-1 shows the assumptions associated with the two different versions of the Fall-BSM model used in the prospective runs. In Fall-D, the *STEP* function is set to 0.0, based on results from the retrospective analysis which found no evidence for a non-zero *STEP* after 1970 (Section 3.3.2). In that case, there is no need to deal further with hypotheses about post-Bonneville mortality of non-transported smolts. If the parameter *D* is specified by the user (Fall-S), then the program will treat *STEP* as an estimable parameter whose retrospective value was non-zero for brood years 1976-1991.

Table 3.2.2-1: Different versions of the Fall-BSM model used for prospective forecasts of Snake River fall chinook.

Model Version	**Model Assumptions**	
	"D"	***"STEP"***
1. Fall-D	selected from the posterior distribution for *D*	*STEP* = 0
2. Fall-S	*D* specified (Table 3.4.1-1)	selected from the posterior distribution for *STEP*, starting in BY 1976

In prospective simulations, we have four basic alternative hypotheses about future values of *STEP*, namely:

- *STEP* = 0 (Full-*D* version in Table 3.2.2-1).
- Regime shift hypothesis, in which *STEP* oscillates in a 60-year cycle between the values of 0.0 (good climatic periods) and a value selected from the posterior distribution for *STEP* (poor climatic periods). The cycle turned non-zero in brood year 1976 (ocean year 1977) (see *Weight of Evidence Report*, Section 4.2.3).
- Hydro-related hypothesis, in which *STEP* will continue in the future at a value selected from it's posterior distribution, assuming a change in brood year 1976, unless the Snake River dams are removed, in which case *STEP* will equal 0.0. This is analogous to the method for spring/summer chinook, described in Appendix H of the *Weight of Evidence Report*, which resolves some of the problems with making post-Bonneville survival proportional to in-river survival (see Section 4.2.3 of *PATH Weight of Evidence Report*). The hypothesis is that the extra mortality was caused by the Snake River dams. With this hypothesis, drawdown of John Day Dam alone would not change extra mortality.
- BKD or "it's here to stay" hypothesis, in which *STEP* will continue in the future at a value selected from it's posterior distribution (see *Weight of Evidence Report*, Section 4.2.3), again assuming a change in brood year 1976.

Ocean Harvest

Ocean harvest in fall BSM is determined by two factors — "maturity schedule" and "ocean harvest schedule." Run reconstructions determined the best estimates of these schedules for past years. During forward simulation for the Snake River stock, the maturity schedule is randomly selected from the period 1964-1991.

The ocean harvest schedule is selected from run years 1985-1996 (the years since the U.S. vs Canada Pacific Salmon Treaty came into effect). The selection of harvest rates is an auto-correlated process because historical harvest rates show an autocorrelation. For example, the age 4 Snake River fall chinook have an autocorrelation of R=0.357 for years 1968-1994. The assumption of a random process is apparently not valid and so a simple autocorrelation was introduced by choosing sequential year harvest rates 50% of the time and random year selection the other 50% of the time. For example, a future simulation might have a selection of harvest rates from year 1990, 1991, 1986, 1992, 1993, 1994, 1988, 1996, etc.

The program allows for changing the ocean harvest levels to be 85% or 115% of the selected schedule. The 15% change in age specific ocean exploitation rates (applied cumulatively) was based on the latest draft of the U.S. proposal to the Pacific Salmon Commission (Draft, February 10, 1998). This change reflects the range of harvest rates used to bracket the relationship between catch and abundance proposed for the managing the major PSC fisheries under an abundance based regime. Note that the proposal is based on impacts to age 4 (adult) fish, but that here we apply the change to all age classes, which likely results in a greater difference from the existing ocean harvest regime. The historical exploitation rates for 1985-1996 are given in Table 3.2.2-2. The base **cumulative** exploitation rates are shown in Figure 3.2.2-1, together with the two optional rates. Policy issues related to the range of alternative harvest scenarios that should be assessed by PATH sensitivity analyses is currently being considered at the policy level by the Regional Forum Implementation Team (I.T.). Future sensitivity analyses could consider a wider range of scenarios, if PATH is instructed to do so by the I.T.

Table 3.2.2-2: Ocean exploitation rates for Snake River fall chinook.

Run Year	Ocean Exploitation Rate				
	Jack	3	4	5	6
1985	0.025	0.105	0.223	0.303	0.357
1986	0.015	0.106	0.170	0.169	0.303
1987	0.037	0.156	0.140	0.159	0.169
1988	0.027	0.060	0.288	0.172	0.159
1989	0.038	0.151	0.233	0.227	0.172
1990	0.042	0.059	0.271	0.252	0.227
1991	0.026	0.051	0.138	0.212	0.252
1992	0.020	0.095	0.242	0.204	0.212
1993	0.006	0.079	0.244	0.204	0.204
1994	0.015	0.014	0.229	0.204	0.204
1995	0.016	0.047	0.074	0.169	0.204
1996		0.046	0.000	0.158	0.169
Mean	0.024	0.081	0.188	0.203	0.219
Min	0.006	0.014	0.000	0.158	0.159
Max	0.042	0.156	0.288	0.303	0.357

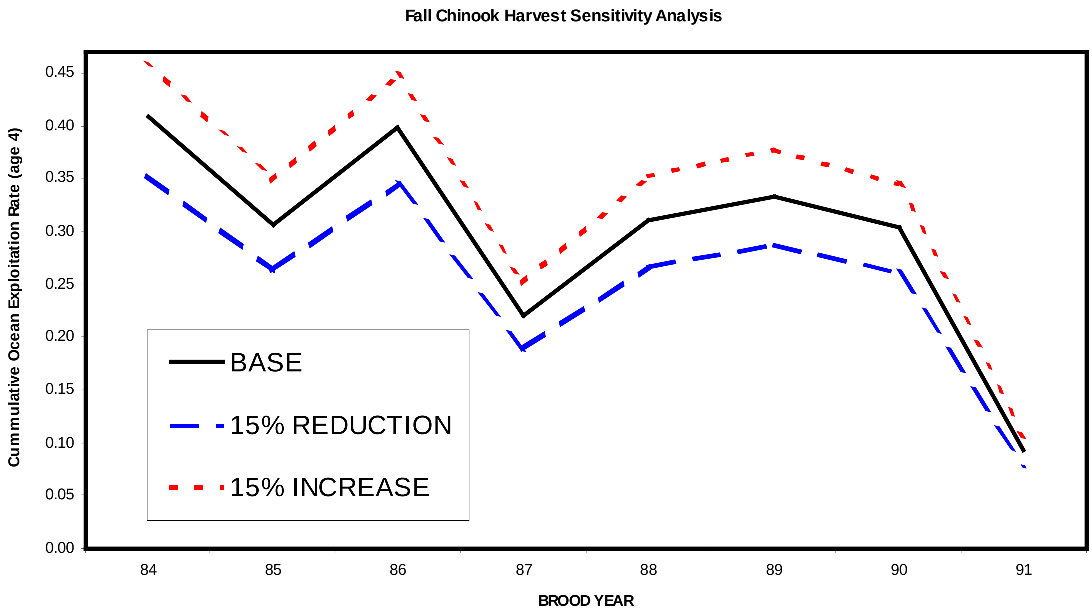


Figure 3.2.2-1: Base *cumulative* ocean exploitation rate (for age 4 fish), and sensitivity analyses conducted above and below this base rate.

In-River Harvest

The current fall BSM in-river harvest rules are based on the 1996-1998 In-River Harvest Agreement[20], modified to reflect current run reconstruction estimates of threshold levels. Since Snake River fall chinook (SRB) enter the river at the same time as the healthy Hanford/Yakima "Upriver Bright" stock (HYURB), both stocks are vulnerable to fishing mortality and the BSM harvest rules are based on the abundances of both of these stocks (Table 3.2.2-3).

Table 3.2.2-3: In-river harvest rates for the SRB and HYURB stocks based on recruits to the river mouth.

HYURB Recruits To River Mouth	**SRB Recruits To River Mouth**							
	0 - 720		**720 - 2,000**		**2,000 - 21,760**		**21,760 +**	
	SRB	*HYURB*	*SRB*	*HYURB*	*SRB*	*HYURB*	*SRB*	*HYURB*
0 -30,000	.07	.09	.07	.09	.07	.09	.07	.09
30,000 – 50,000	.15	.18	.15	.18	.15	.18	.15	.18
50,000 –150,000	.15	.18	.25	.31	.25	.31	.25	.31
150,000 +	.15	.18	.25	.31	.30	.37	.58	.71

The basic idea is that for each combination of recruitment ranges for the SRB and HYURB stocks there is a pair of harvest rates in which the SRB rate is always about 80% of the HYURB rate. These harvest rules are applied in deterministic fashion, based on simulated returns to the mouth of the Columbia River.

[20] 1996-98 Management Agreement for Upper Columbia River Fall Chinook, Washington, Oregon, Idaho, the United States, and Columbia River treaty tribes. *U.S. v. Oregon*, August 10, 1995, Portland, OR

During prospective simulations of a drawdown of the Snake River dams, the threshold levels are further modified to reflect improved up-river survival – the "conversion" rates. The change in threshold is to reduce the SRB thresholds in the first row by 50%, roughly to values, 0, 360, 990, and 10,760. Note that though we vary ocean harvest rates, so far there are no alternative rules for in-river harvest. This should be examined in future sensitivity analyses.

Upriver Survival

Upriver mortality (due to causes other than fishing) between Bonneville and the uppermost Snake River dams are accounted for by using "conversion" rates. These rates include any natural mortality and mortality caused by the four mainstem dams. These are estimated for past years via the run reconstructions (Table 3.1.2-2). In forward simulation these parameters are drawn randomly from the past years 1985-1996 and considered to be representative of current conditions. The conversion rate is applied in conjunction with the in-river harvest rate to get a total in-river survival rate.

To simulate improvements in adult survival during upstream passage due to drawdown, the conversion rate is modified. Table 3.2.2-4 below lists base-level and Snake River dam drawdown scenario conversion rates. The drawdown conversion rates occur in the simulation after dam removal is complete. During the first 2 years of dam removal (construction period) the conversion rates are set to 50% of the base level. One option is given for further improvements in conversion rates for John Day drawdown under Scenario B1: the option boosts conversion rates by 5% above the drawdown rates in Table 3.2.2-4 after John Day dam drawdown – subject to a maximum conversion of 1.0.

Table 3.2.2-4: Snake River fall chinook conversion rates.

Year	Base	Drawdown
1985	0.596	0.975
1986	0.379	0.975
1987	0.376	0.861
1988	0.353	0.975
1989	0.376	0.842
1990	0.378	0.79
1991	0.242	0.732
1992	0.511	0.814
1993	0.56	0.757
1994	0.61	0.782
1995	0.318	0.773
1996	0.367	0.679
Average	**0.422**	**0.830**

Base conversion rates are from Table 3.1.2-2 for Snake River brights (SRB). Drawdown conversion rates are from Table 3.1.2-3 for Hanford-Yakima upriver brights (HYURB; assumed to also represent SRB Bonneville to McNary dam conversion rates), times 0.975 (assumed conversion rate for McNary dam to Lower Granite reach, after drawdown is complete. The 0.975 assumption is based on adult survival studies through free-flowing reaches for spring/summer chinook, which range from 0.95 to 1.0 for this reach.

The difference between the Base and Drawdown columns is quite large in some years, and on average represents about a two-fold increase in upstream survival with drawdown. PATH needs to carefully scrutinize these conversion rates to ensure the projected survival improvement under drawdown is reasonable.

Hatchery Supplementation

Fall-BSM includes a provision for hatchery supplementation. A historical level of hatchery supplementation is selected at random and then added directly to the number of spawners. Thus, supplemented fish are included in assessing the achievement of jeopardy standards. Supplementation is based on random selection of the 1987-1996 Snake River hatchery supplements, which average about 100 fish per year. The current version assumes that hatchery fish spawn and contribute to future recruitment as effectively as natural fish for whatever hatchery supplementation schedule is modeled. Note that the hatchery supplementation represents adult fish that make it back to the spawning grounds, not the number of smolts released. The data for hatchery supplementation are shown below in Table 3.2.2-5. Additions of these supplemented fish can make a substantial contribution to the probabilities of survival and recovery, under all scenarios. PATH needs to assess how sensitive the results are to these supplementation assumptions.

Table 3.2.2-5: Hatchery Supplements

Year	1987	1988	1989	1990	1991	1992	1993	1994	1995	1996
Supplement	48	2	176	74	71	21	152	146	144	172

Carrying Capacity Under Drawdown

The program contains a habitat increase for Snake River stocks under drawdown of the Snake River dams. The increased habitat is modeled by a reduction in the Ricker *b* coefficient by a factor of 1.77 whenever Snake River spawners are above 18,000. The 1.77 factor is based on a 77% increase in potential spawning habitat after drawdown, calculated from the change in length of unimpounded river miles (Earl Weber, unpub.); this indicated a 77% increase. However, the McNary to Lower Granite reach has not generally been used for spawning by Snake River chinook, even prior to the Snake River dams, and is considered to be suboptimal spawning habitat (H. Schaller, pers. comm.). Therefore, we assumed that this reach would only be used for spawning if the population reached higher levels than recorded in the run reconstructions. The highest estimated number of spawners was 17,655 in 1968, which was rounded to 18,000 as an estimate of current carrying capacity.

At present there is no increase in spawning habitat for Snake River fall chinook (SRB) with John Day drawdown, as it is assumed that Snake River fish would proceed upstream to their historic spawning area. John Day drawdown would, however, likely produce an increase in productivity (Ricker *a*) and carrying capacity (Ricker *1/b*) for the Hanford stock (HYURB). At present this change in the Hanford stock has not been implemented, though it will be implemented in future.

References:

Meyers, J.M., R. Kope, G. Bryant, D. Teel, L. Lierheimeyer, T. Wainwright, W. Grant, F Waknitz, K. Neely, S. Lindley, and R. Waples. 1998. Status review of chinook salmon from Washington, Idaho, Oregon, and California. NOAA Tech. Mem. NMFS-NWFSC-35. 318 p. + Appendices. Also see Federal Register Notice, March 9, 1998. Endangered and threatened species: west coast chinook salmon; listing status change; proposed rule. 63 FR 11482.

3.3 Retrospective Results

3.3.1 Passage Model Results, Including Diagnostics

Passage Model Results and Diagnostics

This section describes and compares diagnostic outputs for the two passage models, CRiSP and FLUSH. Although both models draw on common data sets, key differences in input data, functional forms, and modeling structure remain (Table 3.3.1-1). These differences lead to different outputs of the two models, which in turn contribute to differences in the relative outcomes of actions. The purpose of the retrospective diagnostics is to compare and contrast outputs between the two passage models.

Table 3.3.1-1: Major difference between CRiSP and FLUSH retrospective passage models.

CRiSP	FLUSH
• Predator mortality and FTT estimates calibrated to 1995-1997 NMFS PIT-tag survival estimates for Lower Snake River.	• Predator mortality estimates calibrated to 1983-1986 JD Reservoir predation studies.
• Pre-dam survivals derived from model relationships.	• Pre-dam survivals derived from model relationships and second set of runs with fixed survival per mile.
• FTT in Lower Granite pool linearly related to WTT.	• FTT in Lower Granite pool linearly related to WTT with two separate temperature regimes.
• FTT LGR to BON Dam represented with an equation that encompasses a seasonally related FTT/WTT relationship and downstream acceleration.	• FTT from LGR to MCN and MCN to BON represented by reach specific WTT relationships. FTT exponentially related to WTT capped at the FTT 90th percentile.
• Both hatchery and wild PIT-tagged fish were utilized for FTT calibration.	• Wild PIT-tagged fish were used for FTT calibration except in MCN to BON which utilized both hatchery and wild fish.
• Emigration distribution is developed from weighted means of 1991-1997 CPUE distributions of PIT-tagged released wild subyearlings.	• Emigration distribution is developed from 1991-1997 CPUE distribution of ≥ 85 mm PIT-tagged released wild subyearlings with mean release date varying with temperature.
• FTT in LGR reservoir for rearing and migration phase determined from all PIT-tagged hatchery and wild subyearlings released. FTT in LGR for migration only phase assumed the same as LGS FTT.	• FTT in LGR reservoir for migration only phase determined from ≥ 85 mm PIT-tagged wild only subyearlings released. FTT in LGR reservoir for rearing and migration phase determined from all PIT-tagged wild subyearlings released.
• FTT in unimpounded (pre-dam) reaches based on FTT/WTT relationships used in model.	• FTT in unimpounded (pre-dam) reaches set equal to WTT.

The results presented below are for a minimum set of diagnostic measures (survival of non-transported fish (Vn), total survival of transported and non-transported fish to below Bonneville, and proportion of fish below Bonneville that were transported (Pbt)). Some large differences in passage model outputs do exist. However, we have not yet examined these diagnostic outputs in sufficient detail, and have not had time to produce other useful diagnostic outputs, to fully understand the reason for these differences. The fall chinook passage modeling groups are reviewing these diagnostics and are working to understand the differences between the models. In the meantime, we simply compare the outputs of the two passage models and point out some of the major differences in results.

Figure 3.3.1-1 shows retrospective passage model estimates of in-river survival rates of non-transported fish. "Migration only" outputs represent modeled survival of fall chinook smolts from the time they start actively migrating to below Bonneville. "Migration and rearing" outputs represent modeled survival of fall chinook juveniles during the rearing and the migration life stages. The distinction between these two modeling approaches is discussed further in Section 3.4.1. Migration and rearing outputs are lower than those of the migration only outputs, because the migration and rearing also include mortality incurred during the rearing phase. The difference between the two modeling approaches is larger in CRiSP than in FLUSH (Figure 3.3.1-1).

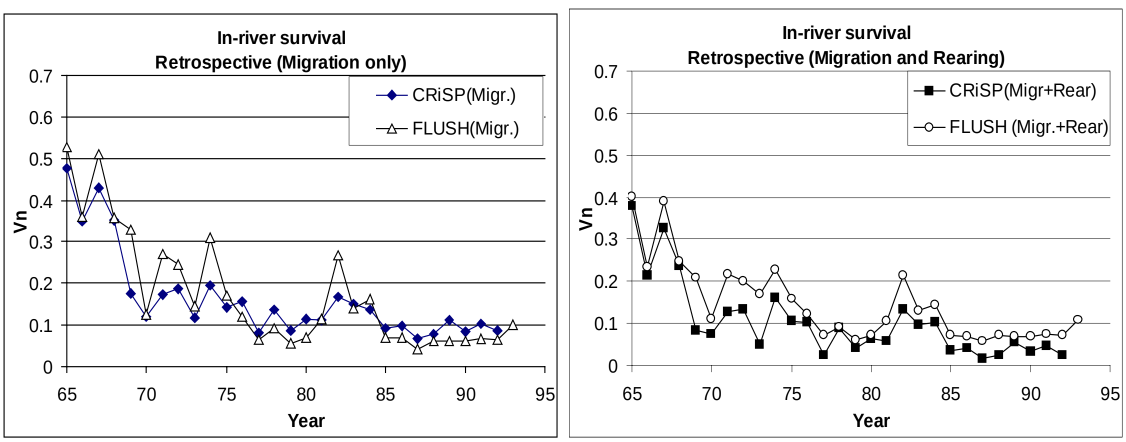


Figure 3.3.1-1: Retrospective estimates of survival of non-transported fish (Vn).

The estimates of in-river survival shown in Figure 3.3.1-1 are for non-transported fish only. Historically, however, a large number of fall chinook smolts have been transported. Figure 3.3.1-2 shows the proportion of fish surviving to below Bonneville (Pbt) that were transported for CRiSP and FLUSH. CRiSP migration and migration+rearing Pbt values were identical. Clearly there are some large differences in the Pbt of the two passage models. Pbt is affected by in-river survival rates, since higher in-river survival rates will lead to more in-river fish surviving to below Bonneville and will therefore reduce the proportion of all fish surviving to below Bonneville that were transported. However, the differences in the in-river survival estimates in Figure 3.3.1-1 do not appear to be large enough to account for the differences in Pbt. Further, with the migration only models CRiSP in-river survival estimates are generally higher than FLUSH and thus would be expected to produce lower estimates of Pbt than FLUSH, all other things being equal.

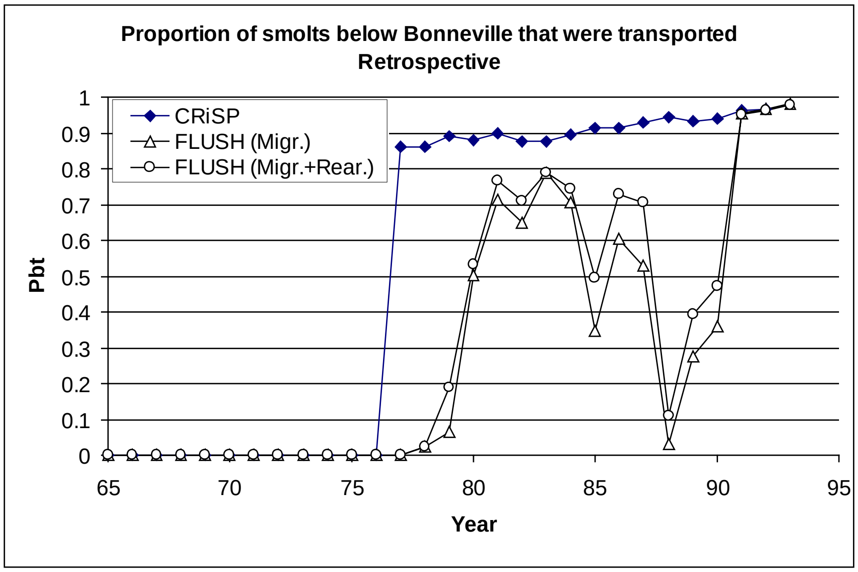
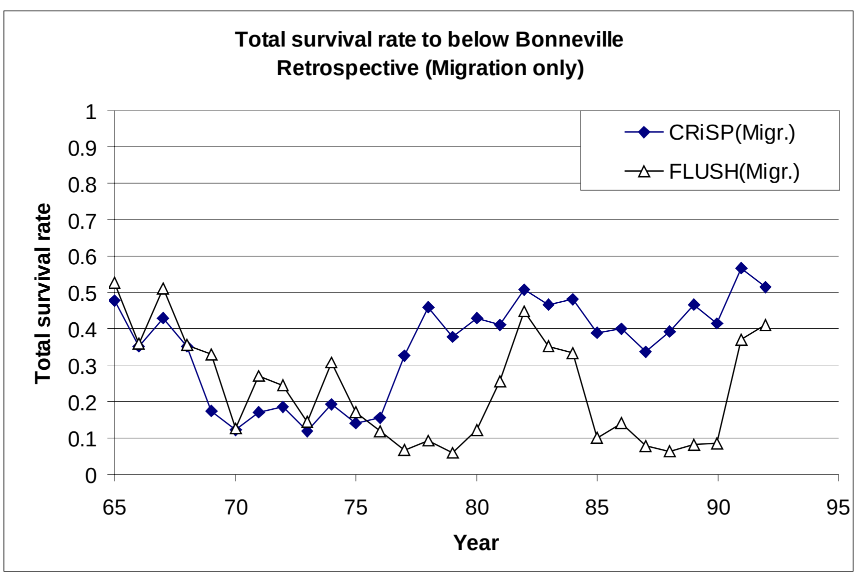


Figure 3.3.1-2: Retrospective estimates of the proportion of fish below Bonneville that were transported (Pbt) CRiSP migration and migration+rearing values are identical.

The two passage models have adopted a value of 0.98 for the survival rate of fish in barges and trucks (see Section 3.1.1). This value can be combined with the in-river survival rate and the proportion of fish arriving below Bonneville to give an estimate of the total survival rate of all fish, transported and non-transported, that arrive below Bonneville (Figure 3.3.1-3). Estimates of the total survival rate from the two passage models are similar with the migration+rearing approach, but are markedly different for the migration only approach. These large differences first appear in 1976, which is when transported started. This suggests that the differences in total survival rate for the migration only models are caused by differences in the Pbt.



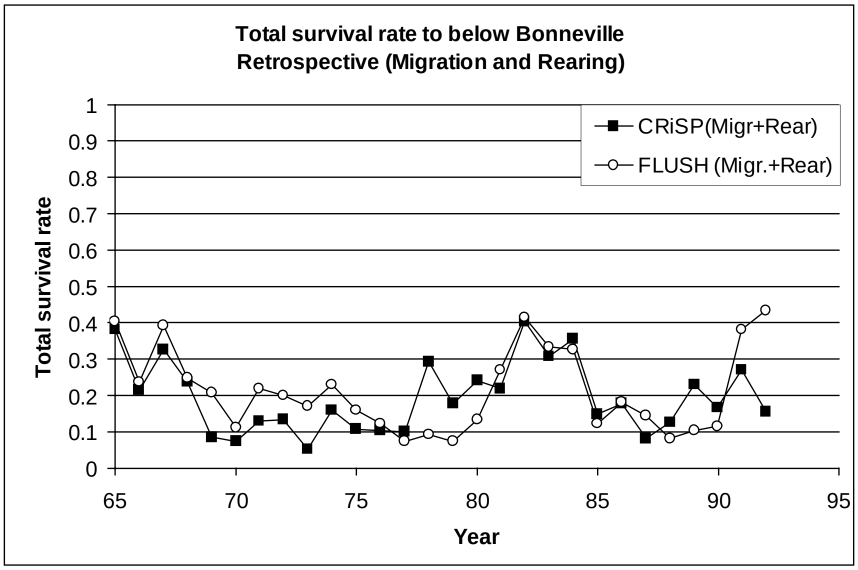


Figure 3.3.1-3: Retrospective estimates of total survival to below Bonneville of transported and non-transported fish.

3.3.2 Life Cycle Results

Retrospective Results

Spawner and recruit estimates are available for the following fall chinook stocks:

- Snake River Brights for brood years 1964-1991
- Hanford Reach-Yakima Upriver natural spawners Brights for brood years 1964-1991
- Lewis River natural spawners for brood years 1964-1991
- Deschutes River for brood years 1977-1991

All stocks have updated run reconstructions based on the ocean exploitation method.

Methods

The emphasis of this retrospective study is the Snake River stock, applying the life-cycle model of Section 3.2.2 to historical spawner/recruit data. The intent of this exploratory analysis was to determine which assumptions are most consistent with these data. We modified the life cycle model in Section 3.2.2 to reflect different combinations of hypotheses, thus creating several different models. The life-cycle model in this application is slightly simplified from the one given in Section 3.2.2 in that there is no depensation included (that is p=0) which is not unreasonable based on results discussed later. Each of these models made predictions of historical recruitment of Snake River fall chinook that were then compared to the field estimates of recruitment. Table 3.3.2-1 describes which combinations of assumptions were run (many other combinations are possible) while Table 3.3.2-2 shows the results, discussed below.

Table 3.3.2-1: Combinations of life cycle model assumptions compared in Table 3.3.2-2. Model codes created from underlined letters or numbers in each row. Shorter codes for models with fewer parameters. BY – brood year.

Full BSM Model # and Code	Passage Model Input	Life History Stages in Passage Model	"*D*" Value Assumed	Starting BY for Estimated "*STEP*"	Year Effect Estimated From
1. CME70DS	CRiSP	Migration only	Estimated	1970	Deschutes-Snake
2. $CMF_0$70	CRiSP	Migration only	Fixed (D=0)	1970	(No Year Effect)
3. CME70	CRiSP	Migration only	Estimated	1970	(No Year Effect)
4. $CMF_{0.37}$70	CRiSP	Migration only	Fixed (D=0.37)	1970	(No Year Effect)
5. CME7OT	CRiSP	Migration only	Estimated	1970	Temperature
6. CME70S	CRiSP	Migration only	Estimated	1970	Spring/Summer
7. CME70P	CRiSP	Migration only	Estimated	1970	PAPA index
8. CME70H	CRiSP	Migration only	Estimated	1970	Hanford stock
9. CM70D	CRiSP	Migration only	Estimated	1970	Deschutes alone
10. Ricker	none	n.a.	(No D)	(No STEP)	(No Year Effect)
11. $CMF_{0.37}$76DS	CRiSP	Migration only	Fixed (D=0.37)	1976	Deschutes-Snake
12. CRE70DS	CRiSP	Migration + Rearing	Estimated	1970	Deschutes-Snake
13. FME70DS	FLUSH	Migration only	Estimated	1970	Deschutes-Snake
14. FME70	FLUSH	Migration only	Estimated	1970	(No Year Effect)
15. $FMF_{0.37}$70	FLUSH	Migration only	Fixed (D=0.37)	1970	(No Year Effect)
16. $FMF_{1.0}$70	FLUSH	Migration only	Fixed (D=1.0)	1970	(No Year Effect)
17. FME70T	FLUSH	Migration only	Estimated	1970	Temperature
18. FME70S	FLUSH	Migration only	Estimated	1970	Spring/Summer
19. FME70P	FLUSH	Migration only	Estimated	1970	PAPA index
20. FME70H	FLUSH	Migration only	Estimated	1970	Hanford stock
21. FME70D	FLUSH	Migration only	Estimated	1970	Deschutes alone
22. Ricker	none	n.a.	(No D)	(No STEP)	(No Year Effect)
23. $FMF_{0.37}$76DS	FLUSH	Migration only	Fixed (D=0.37)	1976	Deschutes-Snake
24. FRE70DS	FLUSH	Migration + Rearing	Estimated	1970	Deschutes-Snake

Table 3.3.2-2: Retrospective analysis of Snake River fall chinook. Model assumptions and codes listed in Table 3.3.2-1. Fixed values of *D* shown in italics (as contrasted with estimated values). Symbols: #parms indicates number of Snake River parameters estimated; "a" indicates Ricker 'a' coefficient; *D* parameter; *STEP* parameter; RSS indicates Snake River residual sum of squares; 2*ln(L) indicates twice the logarithm of the Snake River likelihood; AIC is –2*ln(L) + 2(#parms), the Akaike Criterion; 2ln(Pos) is twice the logarithm of the Snake River likelihood times the prior density of the Ricker "b" parameter; 2ln(J Pos) is twice the logarithm of the joint posterior density for Snake River and Deschutes stocks.

Model #	Model Code	# parms	"a"	*D*	*STEP*	RSS	2*ln(L)	AIC	2ln(Pos)	2ln(J Pos)
CRiSP passage input										
1	CME70DS	5	4.48	0.08	8.4E-06	7.58	-56.72	66.72	-57.06	-105.8
2	$CMF_{0}70$	3	4.99	*0.00*	8.4E-06	10.90	-66.88	72.88	-66.88	-116.3
3	CME70	4	4.69	0.04	8.5E-06	10.23	-65.10	73.10	-65.22	-114.4
4	$CMF_{0.37}70$	3	4.05	*0.37*	3.0E-01	15.55	-76.84	82.84	-78.12	-129.2
5	CME7OT	5	4.61	0.05	8.4E-06	10.13	-64.84	74.84	-65.01	-114.2
6	CME70S	5	4.66	0.06	8.4E-06	9.68	-63.55	73.55	-63.62	-112.7
7	CME70P	5	4.83	0.01	8.4E-06	9.92	-64.25	74.25	-64.44	-113.5
8	CME70H	5	4.50	0.08	8.4E-06	8.90	-61.20	71.20	-61.49	-110.4
9	CM70D	5	4.48	0.08	8.4E-06	7.58	-56.73	66.73	-57.08	-105.8
10	Ricker	2	1.92			12.17	-69.97	73.97	-84.98	-134.8
11	$CMF_{0.37}76DS$	4	4.40	*0.37*	8.3E-01	8.17	-58.81	66.81	-59.28	-108.2
12	CRE70DS	5	4.79	0.19	7.6E-06	10.61	-66.14	76.14	-66.71	-116.1
Flush passage input										
13	FME70DS	5	4.50	0.05	7.6E-05	8.18	-58.85	68.85	-58.92	-107.8
14	FME70	4	4.59	0.01	7.1E-08	9.80	-63.90	71.90	-63.94	-113.0
15	$FMF_{0.37}70$	3	4.30	*0.37*	1.1E-01	16.07	-77.76	83.76	-77.98	-129.3
16	$FMF_{1.0}70$	3	4.27	*1.00*	3.7E-01	24.27	-89.30	95.30	-89.56	-144.1
17	FME70T	5	4.59	0.01	7.1E-08	9.79	-63.88	73.88	-63.91	-113.0
18	FME70S	5	4.59	0.02	7.1E-08	9.79	-63.89	73.89	-63.92	-113.0
19	FME70P	5	4.55	0.01	7.1E-08	9.64	-63.44	73.44	-63.53	-112.6
20	FME70H	5	4.58	0.02	8.0E-10	9.77	-63.81	73.81	-63.84	-113.0
21	FME70D	5	4.51	0.05	8.0E-10	8.23	-59.03	69.03	-59.10	-107.9
22	Ricker	2	1.92			12.17	-69.97	73.97	-84.98	-134.8
23	$FMF_{0.37}76DS$	4	4.30	*0.37*	8.1E-02	11.15	-67.52	75.52	-67.75	-117.6
24	FRE70DS	5	4.51	0.04	9.8E-03	6.95	-54.29	64.29	-54.69	-103.6

This exploratory analysis sought to determine answers to the following questions:

1. Which passage model (CRiSP or FLUSH) appears to provide a better fit to historical recruitment estimates? (model 1 vs. model 13; model 12 vs. model 24; Table 3.3.2-1)
2. Does a migration only or a migration + rearing representation better fit the historical recruitment estimates? (model 1 vs. model 12; model 13 vs. model 24)
3. Do models with input values for "*D*" do as well as models which estimate *D* from the historical recruitment estimates? (model 4 vs. model 3 for CRiSP; model 15 vs. model 14 for FLUSH). We used D=0.37 (an arbitrary choice, ln(0.37)=-1), but roughly representative of the median *D* values in the spring/summer chinook analysis (Section D.7, Appendix D, *PATH*

Weight of Evidence Report). We also explored D=0 (CRiSP model 2), and D=1.0 (FLUSH, model 16), to test out extreme values. Note that the MLE estimate for *D* with FLUSH is very close to zero (model 14).

4. Is it more reasonable to assume a *STEP* decline in recruitment in brood year 1970 (close to start of operation of Snake River dams), or in brood year 1976 (after completion of all Snake River dams; time of ocean regime shift)? (model 1 vs. model 11 for CRiSP; model 13 vs. model 23 for FLUSH) How large is the estimated *STEP* with each model?
5. What is the relative performance of models utilizing different predictors of year effects? (models 1, 5-9 with CRiSP; models 1, 17-21 with FLUSH)
6. How does a simple Ricker model [only "*a*" and "*b*" estimated and no passage model input; model 10/22 (same model)] compare to the other models? The most complex models in this analysis have five estimated parameters (*a*,*b*,*D*, *STEP* and year effect; models 1, 5-9, 12-13, 17-21, and 24).

All but two of the models utilized the "migration only" passage model runs, though other explorations of the "migration + rearing" passage model runs confirm the conclusions described below. Parameter estimates for the models were obtained by either maximizing the posterior density (column 10 of Table 3.3.2-2) or the joint posterior density (column 11). We used the joint posterior density for those models which included the Deschutes residuals in the objective function (Models 1, 11, 12, 13, 23, and 24). Higher values in the two rightmost columns (likelihoods) indicate better fits to the data.

The Akaike Information Criterion (AIC) provides a measure of fit of alternative models, with lower values indicating a better fit. For models with the same number of parameters, a difference in AIC (ΔAIC) < 2 is "insignificant", 2 to 5 is "positive" evidence of a difference in model fit, 5 to 10 is "strong" evidence, and > 10 is "decisive" evidence (Cass and Raftery 1994).

Results

In the following paragraphs, we address the questions raised above, as well as highlight some of the challenges in estimating values for all parameters.

Passage Models

The CRiSP "migration only" model (#1) had a positively better fit (ΔAIC=2.1) than the FLUSH "migration only" model (#13). The FLUSH "migration + rearing + model (#24) had the best fit of any model, with a decisively better fit than the CRiSP "migration + rearing" model (#12); (ΔAIC=11.9).

Within the CRiSP set of runs, the migration only model (#1) showed a strong improvement (ΔAIC=9.4) over the migration + rearing version (#12). Within the FLUSH runs, however, the migration + rearing version showed a positive improvement over the migration-only version (ΔAIC=4.6).

D

Estimated *D* values are generally low (0.04 to 0.19 with CRiSP; 0.01 to 0.05 with FLUSH) unless *D* is taken as a fixed input parameter. Models 4 and 15 set *D* equal to 0.37, an arbitrary choice of a high *D* value. These models did not perform as well as when *D* was estimated from the data (models 3 and 14). These four models all assume that *STEP* starts in 1970.

Akaike criterion scores are generally larger (less favorable) for models where the *D* parameter is set too much larger than the maximum likelihood values. The exception to this conclusion is indicated by model version 11 for the CRiSP passage model, which fixes D=0.37, and assumes that brood year 1976 marks the first year that *STEP* increased in value from a previous value of 0.0. Model 11 has a similar AIC score to model 1 (with an estimated *D* of 0.09 and *STEP* starting in 1970). Under CRiSP, the proportion of smolts transported increases from 0.0 to about 0.86 in brood year 1976 and stays at high values for the remainder of the time series (Figure 3.3.1-3). Such a coincidence of timing between transportation and the onset of extra mortality causes nearly complete confounding of the parameter estimates for *D* and *STEP*. This confounding does not occur to the same extent with the FLUSH passage model; here the proportion of smolts transported increases gradually from 0.0 in 1976 and is more variable throughout the remaining time series. This appears to cause greater discrimination between model 23 (1976 *STEP*; D=0.37) and FLUSH model 13 (1970 *STEP*; estimated D=0.05); model 13 is a strong improvement over FLUSH model 23 (ΔAIC=6.7).

STEP

Post-Bonneville mortality of non-transported smolts is given annually at a rate equal to the parameter *STEP*. We do not know the exact brood year in which this post-Bonneville mortality could have started to increase from a zero rate for the Snake River fall chinook. Two alternative hypotheses were considered: a 1970 and a 1976 brood year initial start of that mortality. The primary run comparisons for the two alternative initial years are given for CRiSP input by models 1 and 11, and for FLUSH input by models 13 and 23. As seen in Table 3.3.2-1, the likelihood, posterior density, and joint posterior density are all maximized with the 1970 initial start year (models 1 and 13) in comparison to an alternative 1976 initial start year with an assumed D=0.37; although see the discussion of potential confounding in the previous section. If 1970 is the initial start year for non-zero *STEP*, or if FLUSH passage model input is used, then the maximum likelihood or posterior occurs with *STEP* essentially at a zero values throughout the time series (Table 3.2.2-2). An advantage of that result is that prospective analyses of future events are greatly simplified because if *STEP*=0 there is no need to deal with alternative hypotheses about the causes of post-Bonneville mortality of non-transported smolts. *STEP* only deviates from a value close to zero when *D* is fixed at 0.37 (models 4, 11, 15, and 23). That is, less post-Bonneville mortality of transported smolts (higher *D*) forces a higher post-Bonneville mortality of non-transported smolt (higher *STEP*) to fit the data.

Year-Effect

Results from models 9 and 21 indicate that the year-effect is best modeled by the residuals of the Deschutes Ricker spawner-recruit model, as described in Section 3.2.2. Models 9 and 21 provide nearly identical results to models 1 and 13, which are fit to the joint distributions of Snake River and Deschutes' residuals. None of the other alternative year-effects improve the fitting of the model nearly as well, as shown in models 5 – 8 and 17 – 20. Models 9 and 21 show positive to strong reductions in AIC scores over these other year-effect models. The alternatives tested included:

a) Temperature index based on Saan-Yoon's thesis; namely the average SST from five Canadian weather stations during the period October - January — those five stations are Langara Island, Cape St. James, Pine Island, Kains Island, and Amphitrite Point (models 5 and 17);

b) MLE estimate of year-effect from the delta model application to spring/summer chinook (models 6 and 18);

c) PAPA index of year-effect from the alpha model application to spring/summer chinook (models 7 and 19); and

d) Residuals of a fitting of the Ricker model to the Hanford fall chinook data (models 8 and 20).

Ignoring the year effect reduces the fit (i.e., model 1 outperformed model 3, and model 13 outperformed model 14).

Simple Ricker Model

Simple Ricker models which leave out any passage model input, *D*, *STEP* and the year effect (models 10/22; these are the same model) did not perform as well as models which included these parameters (models 1 and 13). The AIC differences (#10/20 vs #1 = 7.3; #10/20 vs. #13 = 5.1) were both indicative of strong differences. The large difference in Ricker "*a*" values between model 10/22 and the other models is due to the fact that the direct and delayed mortality terms (*M*, *STEP*) force intrinsic productivity (Ricker "*a*") to a higher level to counteract the added mortality and still fit historical recruitment estimates.

Depensation Parameter p

The retrospective models in Tables 3.3.2-1 and 3.3.2-2 assumed p=0 (no depensation). This assumption was justified because when this parameter was included, the estimated values for p were very close to zero (median of posterior distribution is 10^{-4}, with 90% of the distribution between 10^{-6} and 10^{-2}; Table 3.3.2-3). However, the lack of data at low spawner numbers constrains the model's ability to detect depensatory effects. Therefore, it is important that PATH do further work to check on the validity of estimates of p, and to conduct sensitivity analyses on the effects of different p values on the results.
If depensation were present, it could significantly limit the ability of stocks to recover. PATH could test the ability to detect depensation (i.e., p values greater than zero) by generating hypothetical spawner/recruit data from a simulation model in which depensation is present, and using this as input to the parameter estimation procedure.

Graphs of Best Fitting Models

Figures 3.3.2-1 and 3.3.2-2 show observed and predicted values from application of the fall chinook models to models 1 and 24 – the highest likelihood fits for CRiSP and FLUSH passage model input, respectively. As seen in the graphs, both models capture the general features of the observed data. Figure 3.3.2-3 shows residuals for the ln(R/S) for the four fall chinook stocks – note that the Snake River residuals are from fits to model 24, which is similar to results for model 1. The data show no obvious declines in (R/S) in brood years 1970 to 1976, but do show generally lower values after brood year 1984.

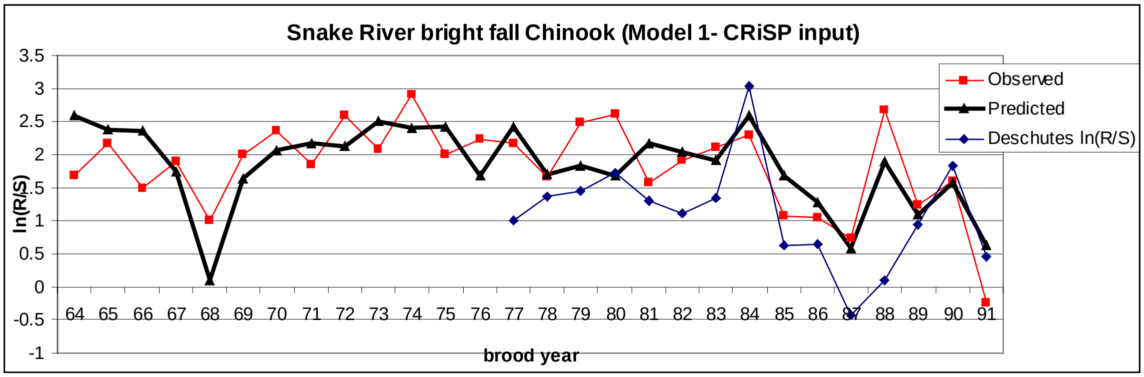


Figure 3.3.2-1: Observed and predicted values from application of the fall chinook models to CRiSP passage model input (model 1 in Table 3.3.2-2).

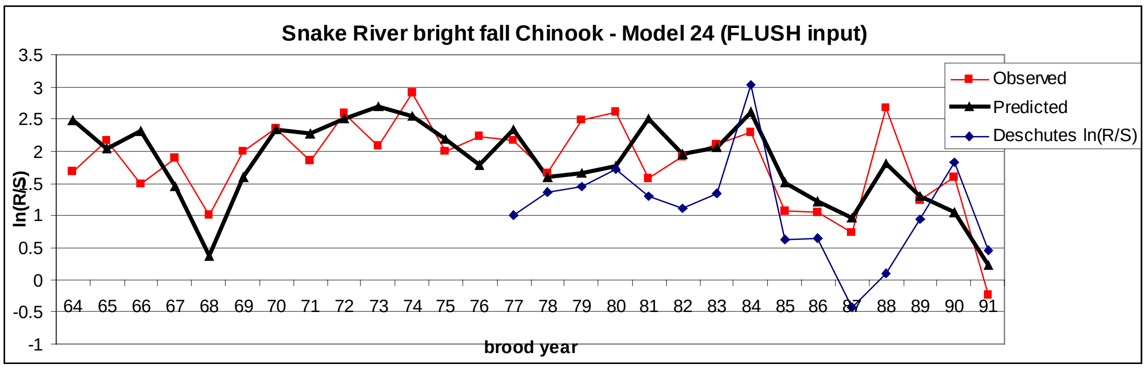


Figure 3.3.2-2: Observed and predicted values from application of the fall chinook models to FLUSH passage model input (model 24 in Table 3.3.2-1).

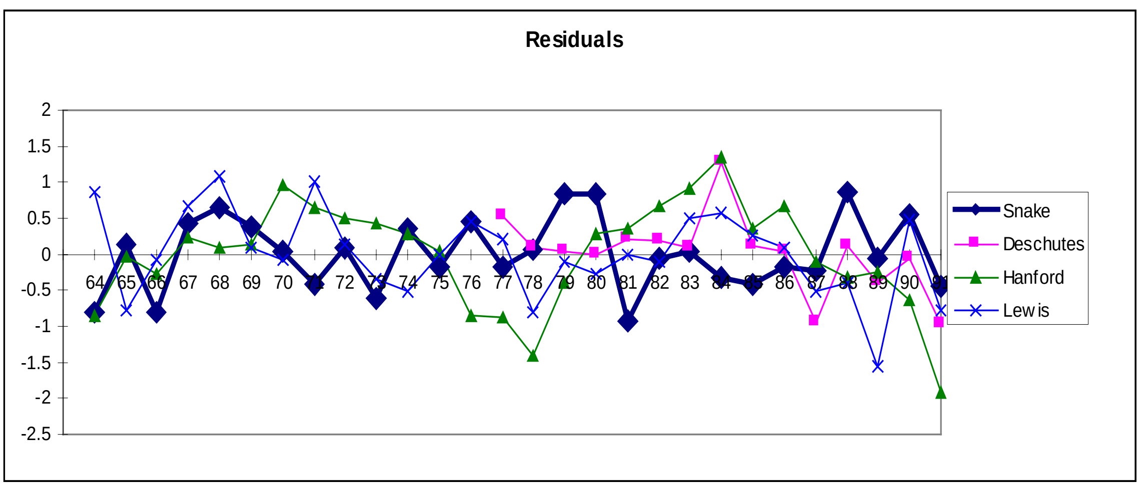


Figure 3.3.2-3: Residuals for ln(R/S), for each of the four fall chinook stocks, based on model 24 (migration + rearing model using FLUSH input, see Table 3.3.2-1).

Posterior Intervals for Model Parameters

After completion of the exploratory analysis of the life-cycle model (Table 3.3.2-2), we finalized the structure of Fall-BSM for prospective analyses. Models 1, 12, 13 and 24 were the basic model forms used for prospective analyses (Table 3.3.2-1), though some changes were made to *STEP* and *D* as outlined below. Fall-BSM takes input from either passage model (CRiSP or FLUSH), in either form (migration only or migration + rearing) and then considers four alternative hypotheses for extra mortality:

1. *D* estimated; *STEP*=0
2. regime shift hypothesis (*D* specified; *STEP* estimated)
3. BKD hypothesis (*D* specified; *STEP* estimated)
4. Hydro hypothesis (*D* specified; *STEP* estimated)

These hypotheses are explained further in Section 3.4.

One of the useful outputs of the Fall-BSM is the posterior intervals for the parameter estimates (i.e., probability distributions for uncertain parameters, based on the fit to historical recruitment estimates). Quantiles of the 10%, 25%, 50%, 75%, and 90% posterior probabilities for the model parameters in models where STEP = 0.0 are provided in Table 3.3.2-3 below (applied in prospective models using the first of the above four extra mortality hypotheses). The complete model in Section 3.2.2 is applied and a parameter transformation of the Ricker "*a*" coefficient was made to improve numerical stability.

Table 3.3.2-3: Posterior probability intervals for model parameters (parameters defined in Section 3.2.2), when *STEP* is set equal to zero, and different passage models are used to estimate in-river mortality (*M_t*) and portion of smolts transported (*Pb_t*). Symbol "C" indicates CRiSP passage model; "F" indicates FLUSH passage model; "mig" indicates a migration only scenario; "mig+rear" indicates a migration + rearing scenario. Percentages are the quantiles for each column.

Passage Model	Parameter	10%	25%	50%	75%	90%
C mig	a+ln(b)	12.21	12.57	13.17	13.88	14.33
C mig+rear	a+ln(b)	12.50	12.87	13.50	14.25	14.74
F mig	a+ln(b)	12.07	12.42	13.01	13.72	14.17
F mig+rear	a+ln(b)	12.25	12.63	13.25	13.95	14.38
C mig	g	0.39	0.59	0.82	1.04	1.24
C mig+rear	g	0.22	0.43	0.66	0.90	1.11
F mig	g	0.25	0.46	0.69	0.92	1.14
F mig+rear	g	0.34	0.55	0.76	0.99	1.20
C mig	ln(b)	-9.55	-9.33	-8.96	-8.48	-8.15
C mig+rear	ln(b)	-9.65	-9.40	-8.98	-8.51	-8.22
F mig	ln(b)	-9.37	-9.14	-8.77	-8.30	-7.99
F mig+rear	ln(b)	-9.57	-9.33	-8.94	-8.46	-8.16
C mig	ln(D)	-3.26	-2.74	-2.26	-1.85	-1.52
C mig+rear	ln(D)	-2.19	-1.81	-1.48	-1.16	-0.88
F mig	ln(D)	-3.97	-3.27	-2.59	-2.04	-1.60
F mig+rear	ln(D)	-4.09	-3.28	-2.66	-2.11	-1.66

Passage Model	Parameter	10%	25%	50%	75%	90%
C mig	ln(p)	-14.82	-12.86	-9.84	-6.55	-4.59
C mig+rear	ln(p)	-14.66	-12.72	-9.37	-6.06	-3.89
F mig	ln(p)	-14.60	-12.53	-9.25	-6.15	-4.07
F mig+rear	ln(p)	-14.75	-12.92	-9.82	-6.72	-4.72
C mig	ln(Var e_t)	-1.09	-0.99	-0.87	-0.75	-0.63
C mig+rear	ln(Var e_t)	-0.98	-0.87	-0.75	-0.62	-0.51
F mig	ln(Var e_t)	-1.06	-0.96	-0.84	-0.71	-0.60
F mig+rear	ln(Var e_t)	-1.12	-1.01	-0.89	-0.76	-0.64

Adjusting D Estimates Through Comparison to Snake River Basin Spring/Summer Chinook

Our analyses for Snake River spring/summer chinook estimated *D* form transport:control studies originating at lower Granite Dam and estimates of in-river survival. However, we do not have transport:control studies for Snake River fall chinook from Lower Granite Dam. As a result, we used a different method: estimating *D* from spawner/recruit data. We do not know if this method causes any biases. Therefore, we applied the spawner/recruit method of estimating *D* values to spring/summer chinook, and compared the estimates we obtained with *D* values derived from transport:control studies and passage models. The limited comparison uses a life cycle model like that described in Section 3.2.2, with a non-zero *STEP* initiated in brood year 1976, and a spring/summer year-effect proportional to the MLE estimates of year-effect from the Delta model (see *PATH FY1997* report). Further simplifications are that only the seven Snake River basin stocks were considered (no down-river stocks) and a single version (TURB 4) of passage model assumptions was employed as input.

Results from this analysis are shown below in Table 3.3.2-4. Three versions of the model were run: 1) estimating both *D* and *STEP*; 2) fixing *D* at the lower 95% confidence interval of the estimates in version 1; and 3) fixing *D* at the upper 95% confidence interval of the estimates in version 1. The *D* values estimated with FLUSH are higher than those estimated with CRiSP, and show a larger range (Table 3.3.2-4). The *D* values in Table 3.3.2-4 are below the median *D* values from passage models and T:C studies, but lie within the range of the passage model values. The median *D* value from FLUSH TURB 4 is 0.336 with a range of (0.11, 1.0) for the 24 values over years 1971-1992; the median *D* value from CRiSP TURB 4 is 0.228 with a range of (0.004, 3.43) over years 1968-1995.

Table 3.3.2-4: Results from applying the spawner/recruit method of estimating D values to Snake River region spring/summer chinook. TURB 4 passage model assumptions were used, and seven Snake River spring/summer chinook stocks. Model versions 2 and 3 indicate approximate 95% confidence intervals for the MLE estimated *D* values in model version 1.

FLUSH

BSM model version	*D*	*STEP*	R^2	RSS	2ln(Likelihood)
1. with both *D* and *STEP* estimated	0.25	3.87E-11	0.51	158.28	-1038.20
2. with *D* fixed at lower 95% C.I.	0.15	4.83E-12	0.50	161.48	-1042.30
3. with *D* fixed at upper 95% C.I.	0.82	7.46E-01	0.50	161.27	-1042.03

CRiSP

BSM model version	*D*	*STEP*	R^2	RSS	2ln(Likelihood)
1. with both estimated	0.102	3.87E-11	0.58	137.18	-1008.86
2. with *D* fixed at lower 95% C.I.	0.050	3.87E-11	0.57	139.87	-1012.85
3. with *D* fixed at upper 95% C.I.	0.198	1.48E-01	0.57	139.88	-1012.86

For purposes of sensitivity analysis, a calculation was made of the difference between the median *D* estimates from passage models and T:C studies, and the MLE *D* estimates in Table 3.3.2-4, using the spawner/recruit data:

CRiSP: median *D* – MLE *D* = 0.126 ≈ 0.13
FLUSH: median *D* – MLE *D* = 0.086 ≈ 0.09

These differences imply that the spawner/recruit based estimates of *D* for fall chinook could underestimate *D* by 0.09 to 0.13, relative to what one would expect from Lower Granite T:C studies (if they existed) coupled with passage model estimates of the survival of control fish. These differences were added to the *D* estimates in Table 3.3.2-2 to account for potential bias, as indicated in Table 3.2.2-5. The D values listed in Table 3.3.2-5 were used in forward simulations.

Table 3.3.2-5: Correction of Snake River fall chinook *D* estimates in Table 3.3.2-2 using the results of method comparison on spring/summer chinook (see text). The adjusted *D*'s in the last column were used in prospective simulations requiring specified *D* values (i.e., regime shift, BKD and Hydro extra mortality hypotheses).

Passage Model (Table 3.3.2-2)	*D* Estimate in Table 3.3.2-2	Assumed Correction	Adjusted *D*
1. CRiSP, migration only	0.08	0.13	0.21
12. CRiSP, migration + rearing	0.19	0.13	0.32
13. FLUSH, migration only	0.05	0.09	0.14
24. FLUSH, migration + rearing	0.04	0.09	0.14

There are several caveats to the limited comparisons made above. First, we chose an arbitrary initial year for the change in *STEP*. Second, there is no time-dependent variation in *D*. In the case of CRiSP passage model input, there is a strong tendency for increased *D* values in recent years for spring/summer Snake

River chinook: the median of the four estimates made after 1980 is 0.633 whereas the median of all studies prior to that is 0.174 (see *Weight of Evidence* Report, Figure D-9, page D-9). In the case of the FLUSH passage model input, there is essentially no trend as the long-term median 0.336 equals the 1980-1992 median *D* estimate. Third, this analysis assumes that difference among the two methods of estimating *D* for spring/summer chinook are directly transferable to fall chinook, despite differences in the life histories of these two groups of salmon and the passage models use for each group.

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Saan-Yoon's thesis

Marmorek, D.R. and C.N. Peters (eds.), J.J. Anderson, L. Barnthouse, L. Botsford, N. Bouwes, P. Budy, T. Cooney, R. Deriso, J. Geiselman, A. Giorgi, R. Hinrichsen, J. Hayes, M. Jones, L. Krasnow, O.P. Langness, D. Lee, C. McConnaha, C.M. Paulsen, R. Peterman, C.E. Petrosky, C. Pinney, H. Schaller, S. Smith, C. Toole, E. Weber, J. Williams, P. Wilson, and R.W. Zabel. 1998. Plan for Analyzing and Testing Hypotheses (PATH): Weight of Evidence Report. Compiled and edited by ESSA Technologies Ltd., Vancouver, BC. 116 pp. and appendices.

3.4 Prospective Results for Fall Chinook

This section provides more details on the relative performance of four Hydro actions under consideration (A2, A2', A3, and B1) for fall chinook. Because most fall chinook are already transported, the A1 option (status quo) is virtually identical to the maximize transportation option (A2), and was not modeled. The organization of this section roughly follows that of the section on spring/summer chinook prospective results. Section 3.4.1 documents the various assumptions and hypotheses investigated by PATH for fall chinook. Section 3.4.2 shows the range, frequency, and summary statistics of preliminary results obtained from these combinations of assumptions (we refer to these combinations as "runs", because each combination represents a distinct and unique run of the life-cycle model). Section 3.4.3 explores the relative effects of the various assumptions and hypotheses on the preliminary results. Section 3.4.4 discusses the implications of the key hypotheses for overall evaluation of actions. We note again that the results presented here are preliminary and have not yet been fully analyzed.

3.4.1 Assumptions and Uncertainties

The set of assumptions and uncertainties incorporated into the decision analysis framework for fall chinook is much smaller than the set investigated for spring/summer chinook. The reasons for this are that the historical data that would be used to formulate and test hypotheses about the response of fall chinook to various factors is much more limited than the data available for spring/summer chinook. In addition, PATH has not had time in FY98 to develop a full suite of hypotheses for fall chinook (hypotheses for spring/summer chinook were developed after a full year of intensive retrospective analyses).

Further retrospective and prospective analyses for fall chinook have been proposed for FY99 to develop and refine hypotheses about the relative effects of various influences on fall chinook populations. For the prospective analyses completed in FY98, PATH has focussed on a set of eight uncertainties and their associated alternative hypotheses (alternative hypotheses for these uncertainties are summarized in Table 3.4.1-1).

1. *In-river survival assumptions* – uncertainty in direct survival of in-river fish, and the partitioning of in-river survival between dam and reservoir survival.
2. *Life-stage modeled by passage models* – fall chinook differ from spring-summer chinook in that their freshwater rearing life stage occurs in the mainstem of the Snake River. This life-stage, in addition to the smolt migration life-stage, is therefore potentially affected by passage conditions and operation of the hydrosystem. For example, the timing of smolitification is dependent on water temperature, which is affected by operation of projects upstream from rearing habitat.

   There are two options for modeling the effects of the hydrosystem on the rearing and migration life-stage of fall chinook. One is to allow the life-cycle model to absorb these effects on the rearing phase into estimates of stock productivity and overall life-cycle survival, and use the passage models to model survival in the migration life-stage. The other option is to use the passage models to explicitly model survival in both the rearing and the migration life-stage. We use both options to investigate their implications for overall results.

3. *Fish guidance efficiency (FGE)* – uncertainty in the effectiveness of extended-length screens in diverting fish away from the turbines, relative to standard-length screens.
4. *Survival outside the hydrosystem* – we have explored four potential factors that affect survival of fall chinook outside of the hydrosystem. Further details about these hypotheses and how they were implemented in provided in Section 3.2.2 of this report.

a) *Transportation* – As with spring/summer chinook, one hypothesis is that the survival of transported fish in the ocean relative to non-transported fish (called the D parameter) is affected by indirect and delayed effects of being transported. For spring/summer chinook, we were able to use Transport:Control estimates from transportation studies to estimate D (see Section A.3.1 of the *Preliminary Decision Analysis* Report for details). However, because similar data do not exist for fall chinook, the relative survival of transported fish must either be externally specified or estimated from spawner-recruit data (see section on implementation below).

b) *Regime shift* – This hypothesis is similar to the regime shift hypothesis for spring/summer chinook (Section A.3.3.3, *Preliminary Decision Analysis Report*), which says that the survival outside of the hydrosystem is related to periodic changes in climatic conditions.

c) *BKD (here to stay)* –Various factors (e.g., incidence of Bacterial Kidney Disease) are hypothesized to have permanent effects on fish populations, regardless of management actions (see Section A.3.3.2 for the analogous extra mortality hypothesis for spring/summer chinook).

d) *Hydro* – This hypothesis says that the factors that affect survival outside of the hydrosytem are related to the hydrosystem, and will persist in the future unless dams are removed.

Implementation in the life-cycle model - Hypotheses b, c, and d are implemented using an extra mortality factor ("STEP") that is estimated by the life-cycle model from spawner-recruit data. Because of limitations in the spawner-recruit data, the relative survival of transported fish must be externally specified when this extra mortality factor is estimated for these hypotheses. For the results in this section, we used the D values in Table 3.3.2-4. These values are based on a comparison of spring/summer D values estimated from spawner-recruit data to values independently estimated from transportation studies.

5. *Harvest scenarios* - Alternative harvest schedules were explicitly considered for fall chinook because total harvest rates (which includes a significant ocean harvest on fall chinook) are higher than those on spring-summer chinook and are therefore potentially a more important factor. We looked at three ocean harvest scenarios: Current, Conservative (0.85 times current rates), and Liberalized (current harvest rates times 1.15).

   The 15% change in age specific ocean exploitation rates (applied cumulatively) was based on the latest draft of the U.S. proposal to the Pacific Salmon Commission (Draft, February 10, 1998). This change reflects the range of harvest rates used to bracket the relationship between catch and abundance proposed for the managing the major PSC fisheries under an abundance based regime. Note that the proposal is based on impacts to age 4 (adult) fish, but that here we apply the change to all age classes, which likely results in a greater difference from the existing ocean harvest regime.

6. *Length of Pre-Removal Period* – the duration of time between a decision to proceed with drawdown and actual removal of dams (pre-removal period) due to uncertainty in the Congressional appropriations process and the possibility of litigation. We used the same assumptions as for spring/summer chinook (3 or 8 years for Snake River dams, 10 or 15 years for John Day dam).

7. *Length of Transition Period* – duration of period between completion of dam removal and establishment of equilibrium conditions in the drawndown section of the river (transition period), reflecting uncertainty in the physical and biological responses to drawdown (e.g., short-term response of predators, release of sediment). We used the same assumptions for fall chinook as for spring/summer chinook (2 or 10 years).

8. *Juvenile survival rate once river has reached equilibrium conditions after drawdown* – uncertainty in the long-term physical and ecological effects of drawdown (e.g., change in density of predators).

For the Snake River dams, we considered two hypotheses intended to bracket the range of possible responses. The upper bound was a survival rate of 0.96, based on per-km survival estimates from the free-flowing Snake R. The lower bound approach was to let the passage models determine survival through the drawndown reach by eliminating direct dam mortality and predation in the forebay and tailraces of the drawndown dams, but to model a faster Fish Travel Time through the drawndown reach.

An upper and lower bound was also modeled for John Day drawdown. The upper bound was 0.98, based on pre-dam per-km survival estimates through the John Day reach. The lower bound was analogous to the lower bound for the Snake River dams – direct John Day dam mortality was eliminated, predation mortality was eliminated in the forebay and tailrace, and Fish Travel Time was reduced through the John Day reach.

9. *Adult conversion rates following John Day drawdown* – uncertainty in the effects of removing John Day dam on the survival of adult fish migrating upstream through the John Day reach.

Uncertainties 6, 7, and 8 only apply when projecting the effects of drawdown to natural river of the four lower Snake River dams (option A3) and of the four Snake River dams + John Day Dam (option B1). Uncertainty 9 was assessed through a sensitivity analysis on a limited set of runs.

Table 3.4.1-1: Set of uncertainties and alternative hypotheses considered for fall chinook.

Uncertainty	Hypothesis Label	Description
In-river survival assumptions —Passage Models	PMOD1	CRiSP estimates of in-river survival (Vn) and proportion transported
	PMOD2	FLUSH estimates of in-river survival (Vn) and proportion transported
Life-stages modeled by passage models	LIFE1	Migration life-stage only
	LIFE2	Migration and rearing life-stages
Fish Guidance Efficiency (FGE)	FGE1	FGE w/ESBS > FGE w/STS (values depend on project) (ESBS = extended length submersible bar screens). (STS = standard length submersible travel screens). e.g., LGR: FGE1 = 49%
	FGE2	FGE w/ESBS = FGE w/STS. e.g., LGR: FGE2 = 53%
	FGE3	FGE w/ SBC (A2' only) e.g. LGR: FGE3 = 61%
Survival outside of the hydrosystem	D	D estimated; extra mortality factor (STEP) = 0.0
	Regime shift	STEP estimated, D value externally specified Extra mortality related to cyclical climatic conditions
	BKD	STEP estimated, D value externally specified Extra mortality here to stay
	Hydro	STEP estimated, D value externally specified Extra mortality here to stay unless dams are removed
Ocean harvest scenario	Current	Current harvest schedule
	Conservative	Current harvest rates X 0.85
	Liberal	Current harvest rates X 1.15
Duration of pre-removal period under drawdown	PRER1	Snake River dams: 3 years John Day dam: 10 years

Uncertainty	Hypothesis Label	Description
	PRER2	Snake River dams: 8 years John Day dam: 15 years
Equilibrated Snake River juvenile survival rate under drawdown	EJUV1	Snake R. and John Day: Survival rate through drawndown reach model-driven, determined by removing forebay and tailrace predation
	EJUV2	Snake R.: Survival rate through drawndown reach = 0.96 John Day: Survival rate through drawndown reach = 0.98
Transition Period: Juvenile survival	TJUVa	Survivals reach equilibrated values 2 years after dam removal (for drawdown of Snake R. and John Day dams).
	TJUVb	Survivals reach equilibrated values 10 years after dam removal (for drawdown of Snake R. and John Day dams).
Adult conversion rate through John Day reach following John Day drawdown (Sensitivity analyses only	No increase	No increase in adult conversion rates through John Day reach following drawdown of John Day dam.
	Increase	Adult conversion rates through john Day reach increase to 1.0 following John Day drawdown.

3.4.2 Range and Distribution of Results

Measures of Performance

The primary performance measures for fall chinook are the NMFS jeopardy standards, calculated as described for spring/summer chinook in Section 2.2.2 of this report. The only difference is that the Snake River fall chinook stock has its own survival and recovery escapement threshold (300 for survival, 2500 for recovery).

As with spring/summer chinook, we present results in terms of the range of outcomes for all runs[21] for a given action and standard, and two summary statistics: an average jeopardy probability for each action and standard, and the fraction of all the runs for a given action that meet the jeopardy standards. Refer to Figure 2.2.2-1 and accompanying text for an explanation of how these statistics were derived. The average and fraction of runs statistics are reported in Section 3.4.4.

Range of Survival and Recovery Probabilities

Cumulative frequency distributions for fall chinook results are shown in Appendix C. Maximum and minimum values for the survival and recovery standards are summarized in Table 3.4.2-1. A2 and A2' produce similar results, as do A3 and B1. For A2 and A2, jeopardy probabilities range from around 0.6 to 1.0 for the survival standards, and from 0.06 to 0.90 for the recovery probabilities. A3 and B1 produce survival standards that range between .83 to 1.0 (for the 24-year survival and recovery probabilities), and between 0.96 to 1.0 (for the 100-year survival and 48-year recovery standards).

Table 3.4.2-1: Minimum and maximum jeopardy probabilities for fall chinook.

Action	24-year Survival	100-year Survival	24-year Recovery	48-year Recovery
A2	0.69 to 1.0	0.65 to 1.0	0.06 to 0.82	0.07 to 0.82
A2'	0.67 to 1.0	0.61 to 1.0	0.06 to 0.90	0.08 to 0.87
A3	0.84 to 0.99	0.96 to 1.0	0.83 to 1.0	0.99 to 1.0
B1	0.84 to 0.99	0.96 to 1.0	0.85 to 1.0	0.99 to 1.0

[21] There 96 runs each for A2 and A2', 384 runs for A3, and 576 runs for B1.

3.4.3 Relative Effects of Hypotheses on Outcomes

We again use Categorical Regression Trees (CART) to identify the hypotheses that have the most influence on outcomes. CART trees for each of the four jeopardy probabilities are shown in Figure 3.4.3-1 to 3.4.3-4. Hypotheses that are split at the top of the tree have greater effects on outcomes than hypotheses that are split at the bottom of the tree. The length of the vertical branches are proportional to the amount of variation in the results that a particular hypothesis explains. Further explanation of how these trees are generated is provided in Section 2.2.3 of this report.

The results of these analyses are similar to the results for spring/summer chinook. They show that the actions themselves are the most important influence in determining the 100-year survival and the recovery standards. The effects of the actions are particularly strong with the recovery standards. A2, and A2' tend to produce lower responses in these jeopardy probabilities, while A3 and B1 tend to produce higher responses. Interestingly, the actions generally account for a small proportion of the variance in the 24-year survival probabilities. For some branches, the effects of the actions are not significant at all. This means that in the short term, assumptions about how the system behaves have a greater effect than the choice of action in determining population levels. Of the alternative hypotheses, the passage models, the different life-stage assumptions modeled in the passage models, and the extra mortality hypotheses tend to account for the largest amount of the differences in results. The most important distinction in the extra mortality hypotheses was the distinction between the *D* hypothesis (*D* estimated, *STEP* set to 0) and the other three hypotheses where *STEP* was estimated and *D* specified (i.e., BKD, Regime Shift, and Hydro). Distinctions between these three hypotheses generally accounted for little of the variance in the results.

3.4.4 Implications for Evaluation of Actions

The results in the previous section provide a focus for moving forward with a weight of evidence type of process for key hypotheses for fall chinook. However, PATH has not yet begun such a process for fall chinook, although this has been proposed for FY99. Until such a process is complete, we are unable to apply anything but equal weights to the alternative hypotheses to calculate the two summary statistics (i.e., all combinations contribute equally to the overall average and fraction of runs meeting the standards). Again, we point out that the results for fall chinook are preliminary and have not yet been thoroughly assessed by PATH.

Average jeopardy probabilities for the actions and standards are summarized in Figure 3.4.4-1. Calculation of these averages is as described in section 2.2.4, except that equal weights were used. Average survival standard probabilities are high (between 0.9 and 1.0) for all actions. However, there is a marked difference in average outcomes with the recovery probabilities. For A2 and A2', average results for these standards are around 0.3, while A3 and B1 produce average results at or near 1.0.

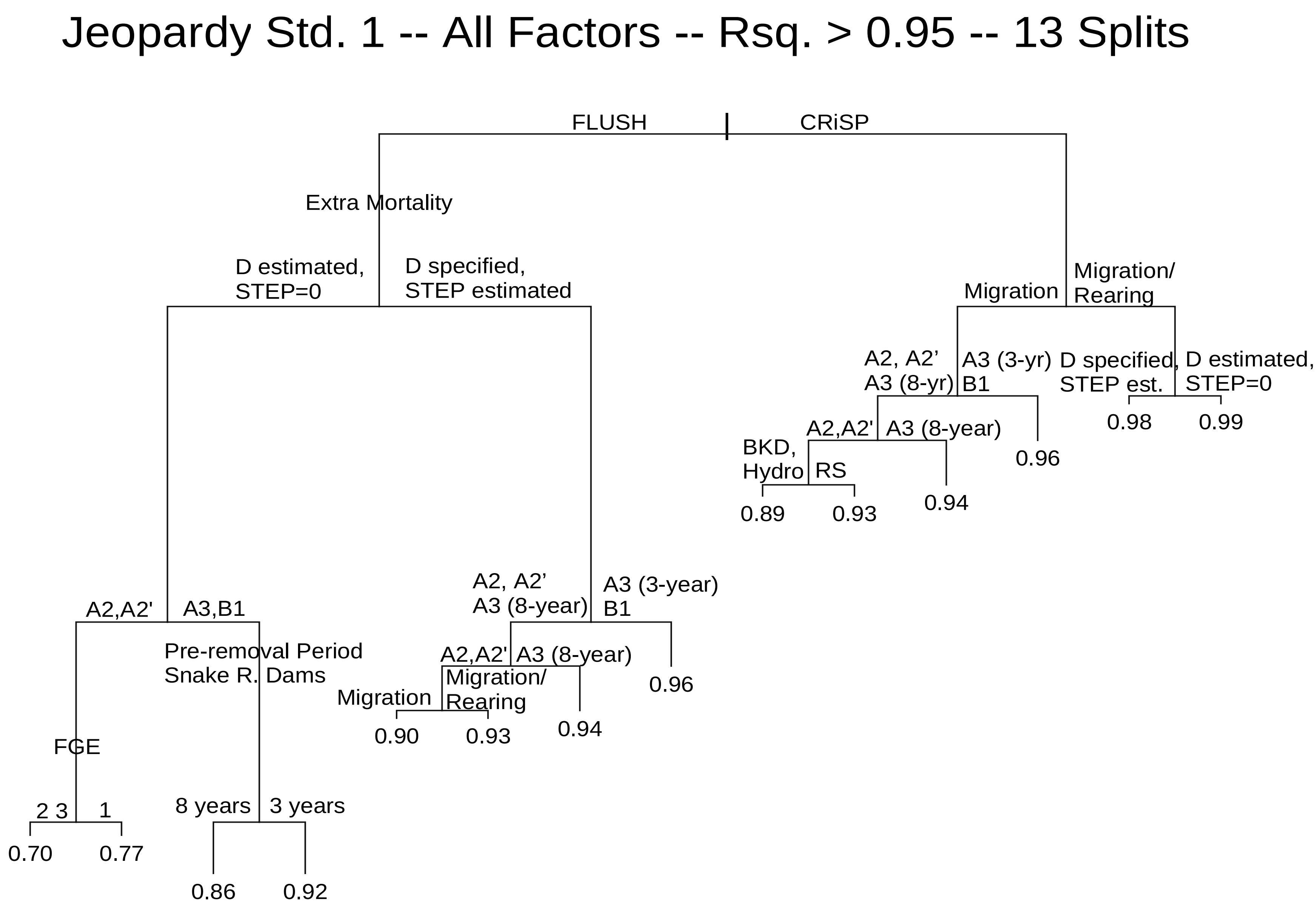

Figure 3.4.3-1: CART diagram for 24-year survival probabilities for fall chinook

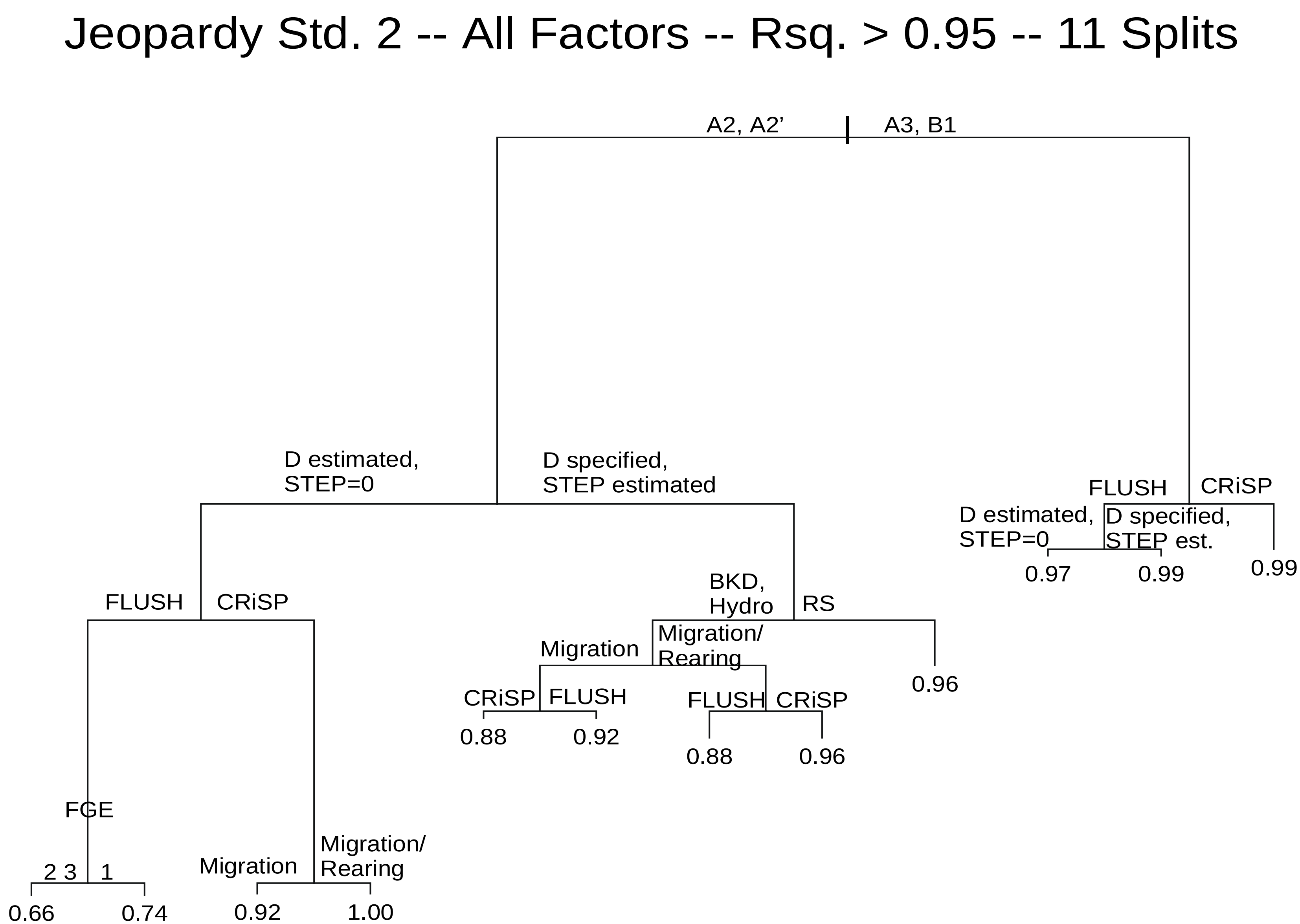

Figure 3.4.3-2: CART diagram for 100-year survival probabilities for fall chinook.

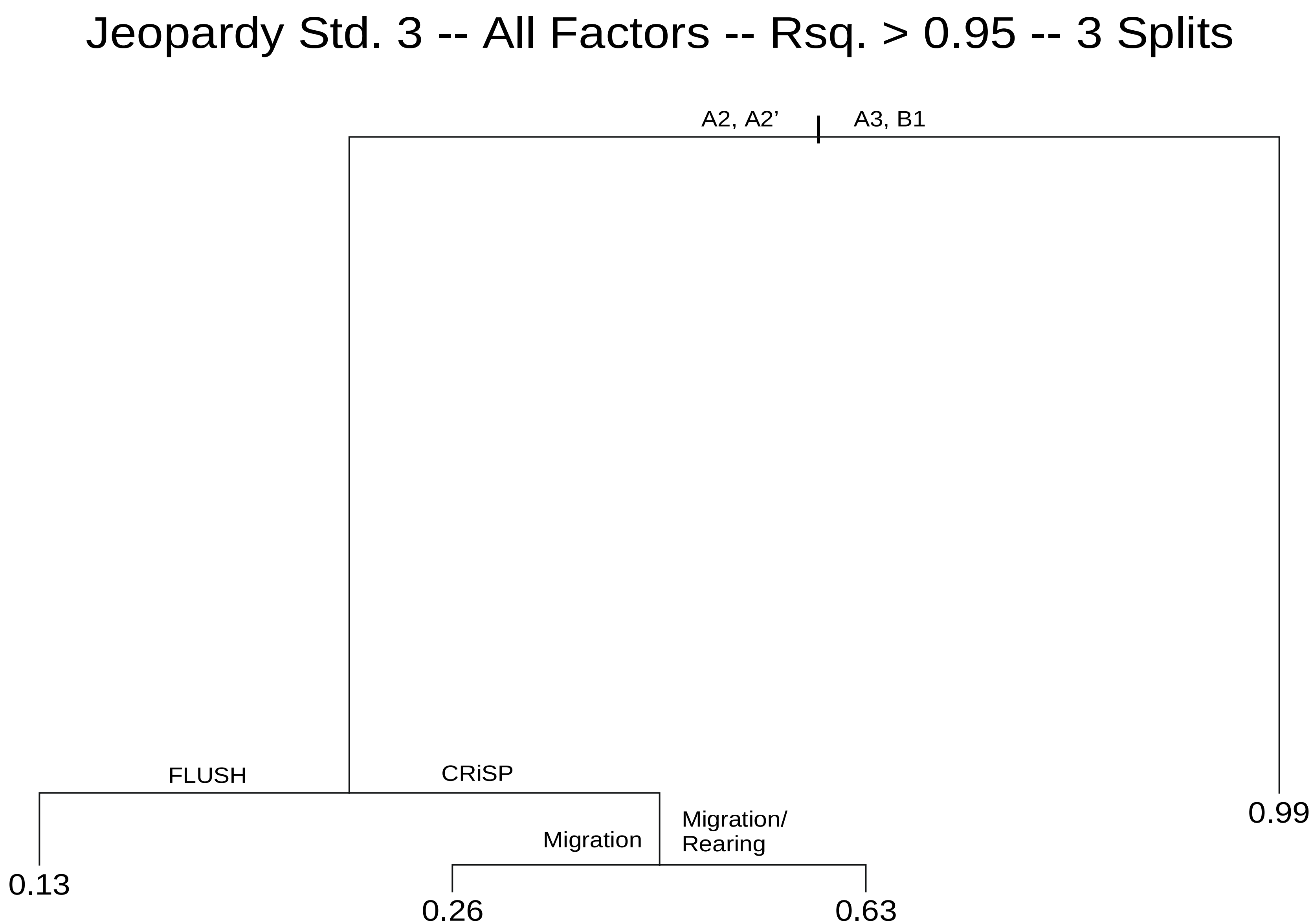

Figure 3.4.3-3: CART diagram for 24-year recovery probabilities for Fall Chinook.

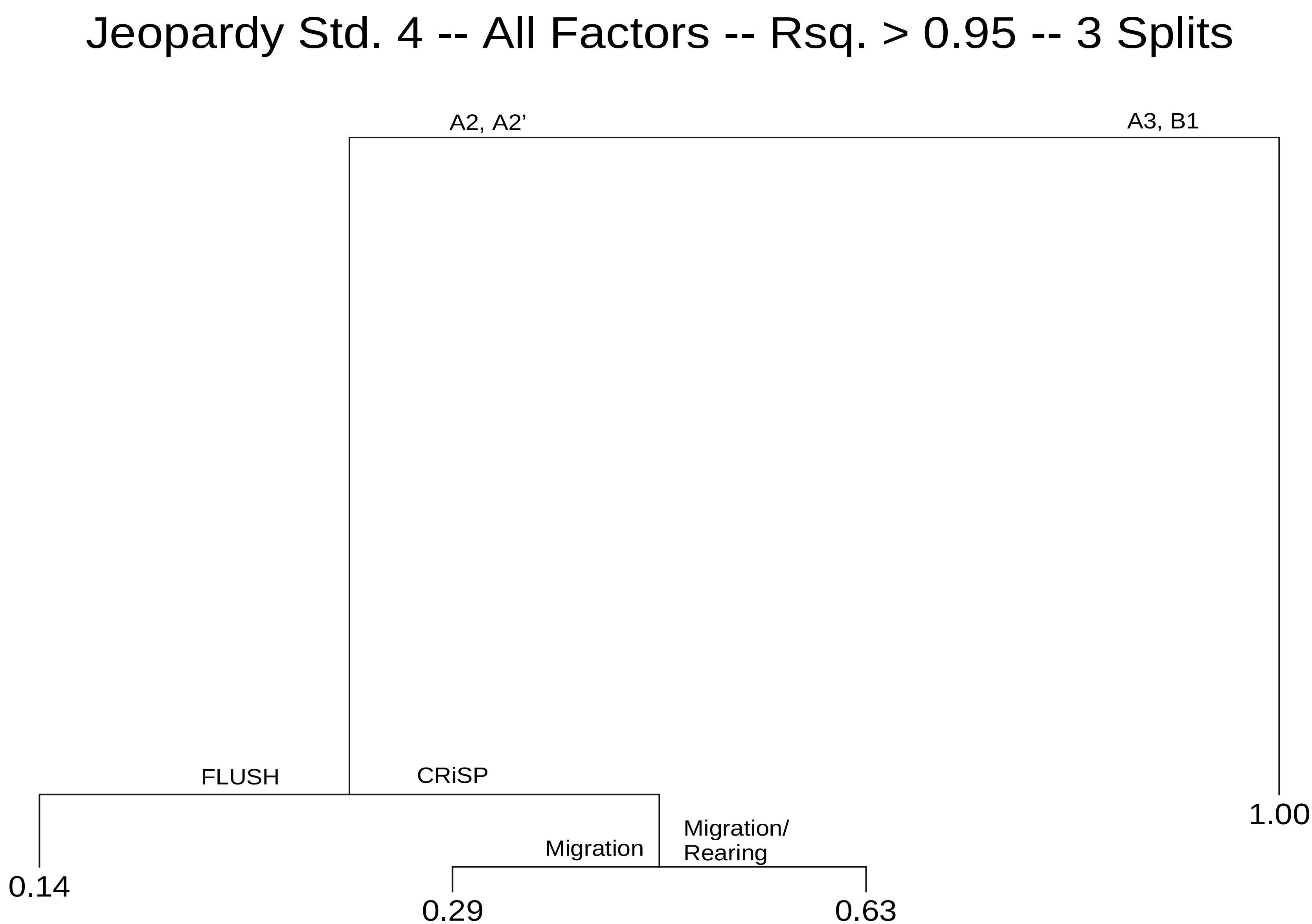

Figure 3.4.3-4: CART diagram for 48-year recovery probabilities for fall chinook.

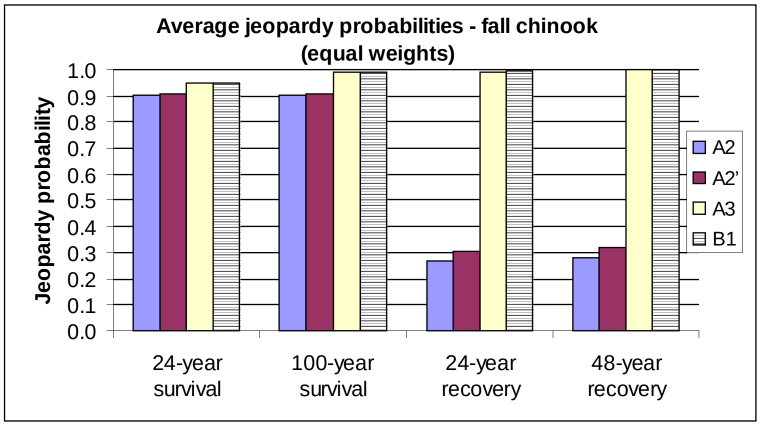

Figure 3.4.4-1: Weighted average jeopardy probabilities for A2, A2', A3 and B1, using equal weights on all hypotheses.

The other summary statistic we present is the fraction of all of the runs for a particular action that equal or exceed the standards defined by NMFS (0.7 for survival, 0.5 for recovery). Figure 2.2.2-1 illustrates how this statistic is calculated. For fall chinook, we use equal weights on all combinations of hypotheses.

Fraction of runs that meet each of the standards is summarized in Table 3.4.4-1. Virtually all A2 and A2' runs meet the 24-year and 100-year survival standards, while 0.15 (for A2) or 0.23 (A2') of the runs meet the 48-year recovery standard. All A3 and B1 runs meet all of the standards.

Table 3.4.4-1: Fraction of runs meeting survival and recovery standards.

	Fraction of Runs Exceeding the Jeopardy Standards		
Action	**24-year Survival**	**100-year Survival**	**48-year Recovery**
A2	0.99	0.95	0.15
A2'	0.96	0.94	0.23
A3	1.0	1.0	1.0
B1	1.0	1.0	1.0

Figure 3.4.4-2 shows the fraction of runs for each action that meets all of the standards (24-year survival, 100-year survival, and 48-year recovery – the 24-year recovery is not an official standard). Section 2.2.4 describes how this statistic was calculated. For fall chinook, we used equal weights. Results mirror the results for the 48-year recovery standard in Table 3.4.4-1. Because virtually all of the runs meet the survival standards, the limiting factor in determining whether all of the runs meet the standards is the 48-year recovery standard.

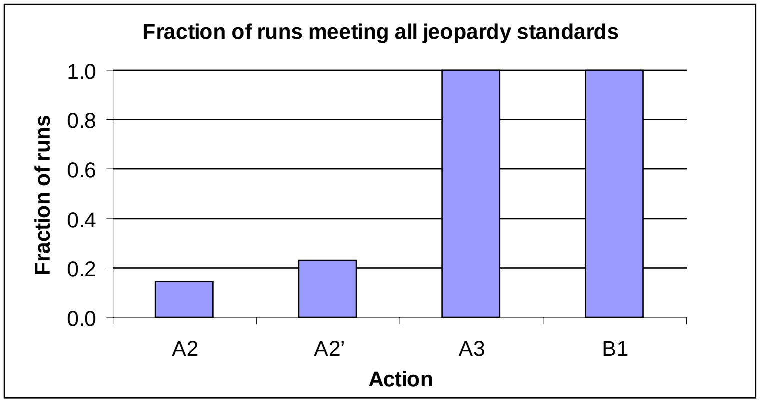

Figure 3.4.4-2: Fraction of runs meeting all of the jeopardy standards (24 and 100-year survival, and 48-year recovery).

These figures show overall results, but effectively mask the effects of alternative hypotheses on the weighted average jeopardy probabilities and fractions of runs meeting the standards. This is particularly important to show for A2 and A2'. Because not all of the runs for these actions meet all of the standards, it is important to know under what sets of hypotheses all of the standards are achieved. Further, we focus on the 48-year recovery standard because all actions are able to meet the 24-year and 100-year survival standards (Table 3.4.4-1). With A3 and B1, all of the runs meet all of the standards, so the effects of particular sets of hypotheses are less important.

Figure 3.4.4-3 compares weighted average 48-year recovery probabilities for A2 only (results for A2' are very similar), for alternative hypotheses about the four major non-drawdown uncertainties (other hypotheses are weighted equally in these results):

a) harvest (B=base harvest scenario; C=conservative harvest; L=liberal harvest)

b) passage model/life stage (C=CRiSP; F=FLUSH; M=migration; R = migration and rearing)

c) extra mortality (D = estimated D, STEP=0.0; R = regime shift; B = BKD, H = hydro)

Passage model and life-stage uncertainties are modeled together to show potential interaction effects between these two uncertainties. Results suggest that the harvest uncertainties have minor effects on 48-year recovery standards for A2, while extra mortality and passage model/life stage uncertainties have larger influences. In particular, modeling both migration and rearing with the CRiSP model generates average 48-year recovery probabilities that are close to the 0.5 standard (58% of the A2 runs and 83% of the A2' runs with this particular set of assumptions achieve all of the jeopardy standards).

Sensitivity analyses on various assumptions made in the fall chinook analyses have not yet been developed to the same extent as for spring/summer chinook. However, we did look at the effects of assuming a 5% increase in conversion rates for Snake River fall chinook following John Day drawdown. The additional 5% was added to the base values listed in Table 3.2.2-4, subject to a maximum conversion rate of 1.0.

Boosting conversion rates by an additional 5% had no effect on results. The largest change in the probability of exceeding the survival or recovery spawning threshold was 0.01. Given that the fraction of fall chinook B1 runs meeting all of the standards is already 1.0 (Table 3.4.4-1), this change in conversion rates will not affect the fraction of runs meeting the standards.

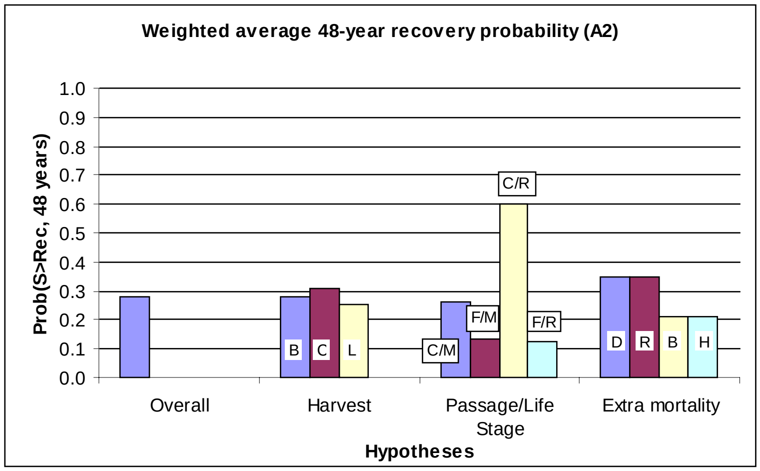

Figure 3.4.4-3. Average 48-year recovery probabilities for A2 with alternative harvest, passage model/life-stage, and extra mortality hypotheses. Codes are defined in the text.

Sensitivity analyses on various assumptions made in the fall chinook analyses have not yet been developed to the same extent as for spring/summer chinook. However, we did look at the effects of assuming a 5% increase in conversion rates for Snake River fall chinook following John Day drawdown. The additional 5% was added to the base values listed in Table 3.2.2-4, subject to a maximum conversion rate of 1.0.

Boosting conversion rates by an additional 5% had no effect on results. The largest change in the probability of exceeding the survival or recovery spawning threshold was 0.01. Given that the fraction of fall chinook B1 runs meeting all of the standards is already 1.0 (Table 3.4.4-1), this change in conversion rates will not affect the fraction of runs meeting the standards.

3.4.5 Comparison of Fall Chinook Passage Survival Measures

As noted earlier, the fall chinook passage modelers have not yet closely scrutinized the outputs of the passage models. A minimal set of retrospective passage model outputs were displayed in Section 3.3.1. Here, we compare prospective passage model outputs for A2, A2’, A3, and B1. As in Section 3.3.1, the purpose is to compare the outputs of the two passage models and highlight some important differences that need further analysis.

In-river survival estimates (Vn) are shown in Figures 3.4.5-1 (A2), 3.4.5-2 (A2’), 3.4.5-3 (A3 and B1, using the lower bound equilibrated juvenile survival rate), and 3.4.5-4 (A3 and B1, with the upper bound assumption for equilibrated juvenile survival rate). See Table 3.4.1-1 and accompanying text for an explanation of the upper and lower bounds. In general, CRiSP Vn’s are higher than FLUSH for A2 and A2’, using the migration only assumption. FLUSH Vns tend to be higher than those of CRiSP for the drawdown actions (A3 and B1), and with the migration + rearing option for A2 and A2’. Vns for A2 and A2’ are approximately the same (0.05 to 0.3), while Vns for A3 are around 0.2 to 0.5, while B1 are Vns around 0.3 to 0.6. The migration vs. migration + rearing distinction appears to have more effect on Vns for A3 and B1 than the upper vs. lower bound equilibrated juvenile survival rate. With FLUSH, the upper and lower bounds on equilibrated juvenile survival rate have virtually no effect on Vn.

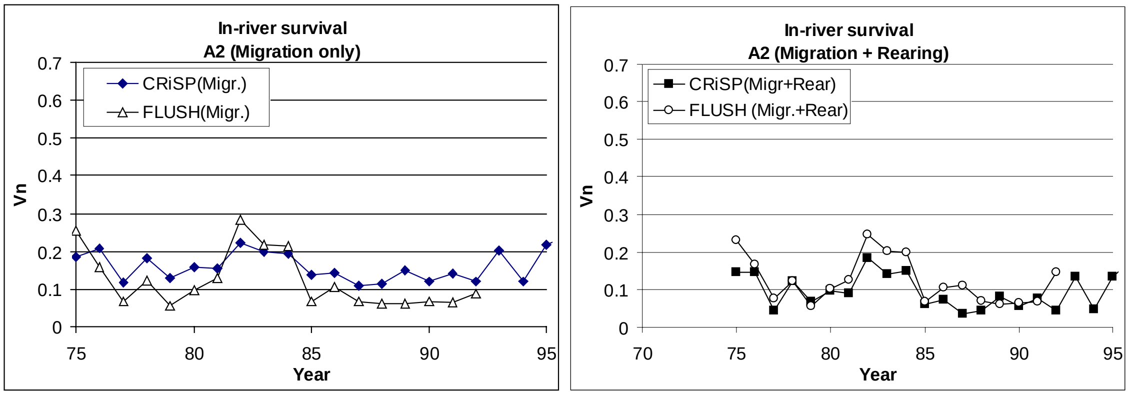

Figure 3.4.5-1: Fall chinook in-river survival estimates for A2.

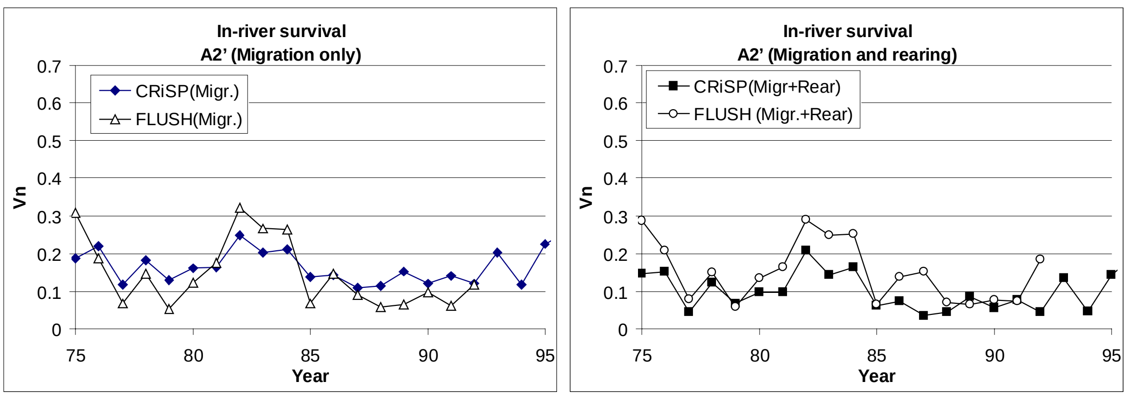

Figure 3.4.5-2: Fall chinook in-river survival estimates for A2'.

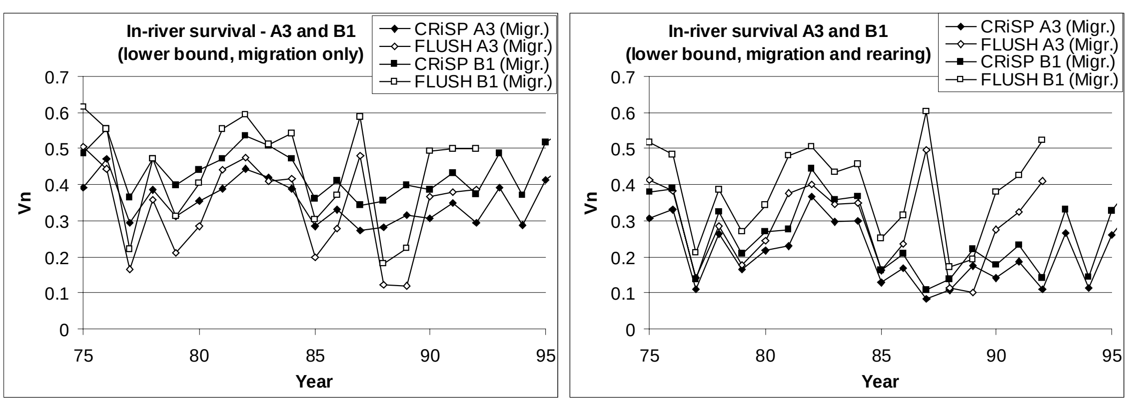

Figure 3.4.5-3: Fall chinook in-river survival estimates for A3 and B1 (using the lower bound assumption for equilibrated juvenile survival rate).

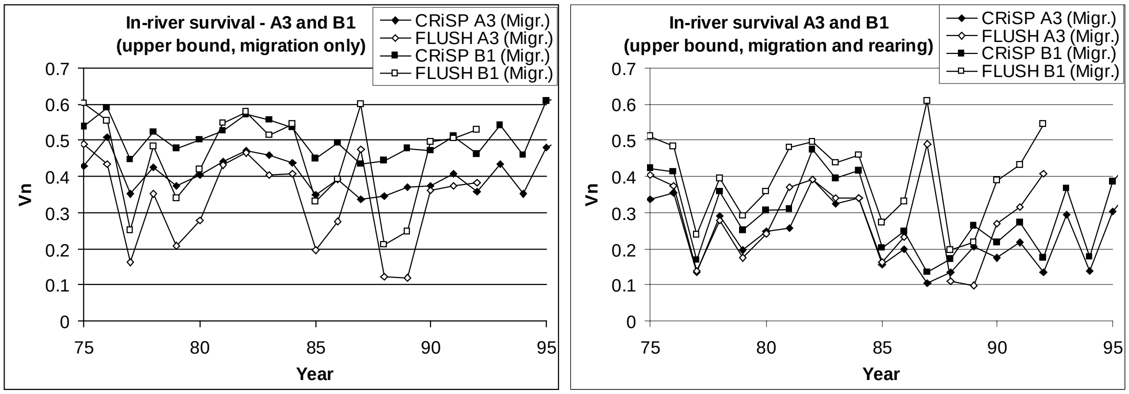

Figure 3.4.5-4: Fall chinook in-river survival estimates for A3 and B1 (using the upper bound assumption for equilibrated juvenile survival rate).

Total survival rates of transported and non-transported fish for A2 and A2' are shown in Figures 3.4.5-5 to 3.4.5-6 (total in-river survival for A3 and B1 is equivalent to the in-river survival because there is no transportation in the drawdown scenarios). Total survival rates for A2' are slightly higher than for A2, and are considerably higher for CRiSP than for FLUSH with migration only assumptions (around 0.1 to 0.3 higher). Proportion of fish surviving to below Bonneville that were transported (Pbt), which also effects total survival, is close to 1 for both CRiSP and FLUSH, which suggests that the differences in total survival are due to differences in the in-river survival estimates produced by the two models.

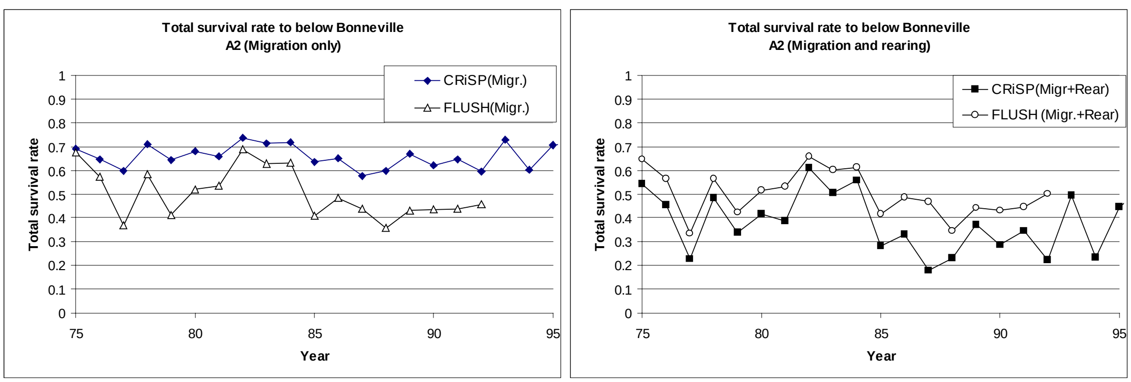

Figure 3.4.5-5: Fall chinook total survival estimates to below Bonneville for A2.

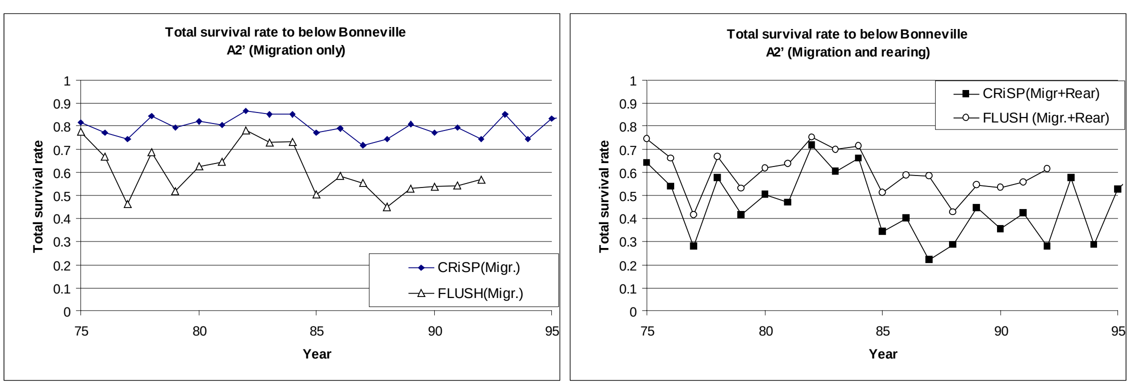

Figure 3.4.5-6: Fall chinook total survival estimates to below Bonneville for A2'.

Finally, we show system survival estimates in Figures 3.4.5-7 (CRiSP) and 3.4.5-8 (FLUSH). System survivals aren't strictly a passage model output, since they involve the use of D values, but they provide a measure of direct + indirect survival of fish through the hydrosystem. System survivals for A2, A3 (both upper and lower bound assumptions for equilibrated juvenile survival rate), and B1 (upper and lower bounds). A2's is not shown because the system survivals are virtually identical to those of A2. We include retrospective system survivals on these graphs for comparison.

Retrospective system survivals are near 0.1 for all models and life-stage assumptions. A2 produces system survival that are near (for CRiSP) or below (for FLUSH) retrospective values. A2 system survivals are less variable than for other actions, because the proportion transported in this scenario is relatively constant from year to year. A3 system survivals range from around 0.1-0.2 (CRiSP, lower bound, migration and rearing) up to 0.4-0.5 (FLUSH, migration only). B1 system survivals are in the range of 0.2-0.4 (CRiSP, lower bound, migration and rearing) to 0.3 to 0.6.

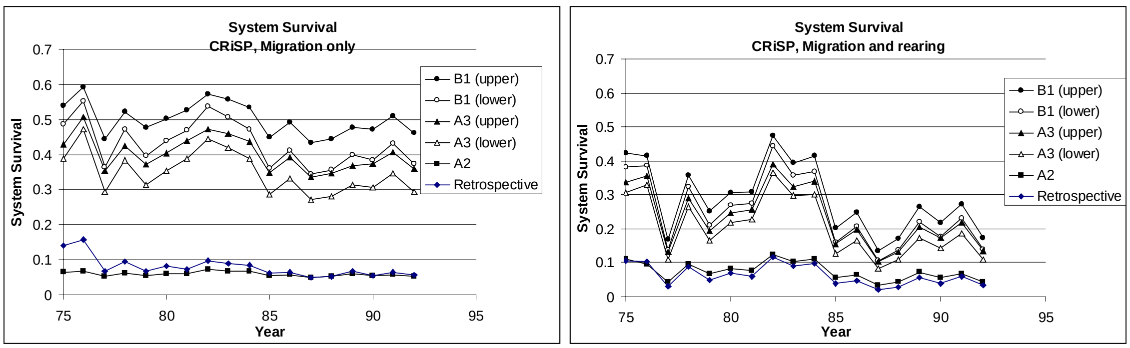

Figure 3.4.5-7: Fall chinook system survival estimates (CRiSP).

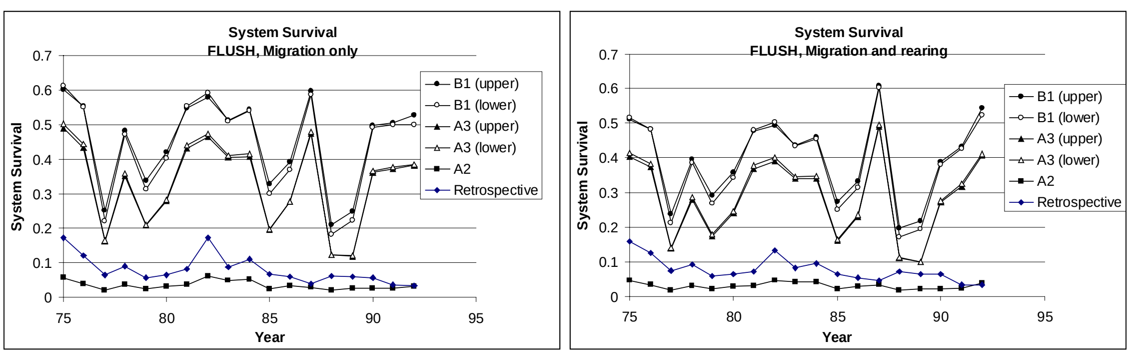

Figure 3.4.5-8: Fall chinook system survival estimates (FLUSH).

4.0 Analysis of Effects of Proposed Actions on Snake River Steelhead

4.1 Introduction

An analysis of the effects of proposed hydrosystem operations on Snake River steelhead is not possible to perform in the same manner as the PATH Snake River spring/summer chinook analysis at this time. The spring/summer chinook analysis is based on a long time series of spawner abundance estimates for populations within the aggregate evolutionarily significant unit (ESU), which can be used to develop run reconstructions and population-level stock-recruitment relationships. Snake River steelhead cannot be effectively censused on the spawning ground except in a limited number of smaller tributaries, so the only available time series is escapement of the aggregate ESU above the upper dam (1962-present). The PATH Scientific Review Panel has stated that modeling an aggregate ESU as if it were a single population is inappropriate because each ESU is actually composed of different populations with different productivities and the mixture will not respond in the same way as an aggregate (Barnthouse et al. 1994). Spawner escapement time series are also nearly non-existent for down-river steelhead stocks, which makes a “delta model” stock contrast approach to the analysis impossible, even if the Snake River ESU were modeled as an aggregate.

A second problem is the lack of performance standards comparable to those available for Snake River spring/summer chinook. Definition of species-level biological requirements for Snake River chinook salmon required the estimation of the “threshold” stock-specific spawner-abundance levels required for continued survival of at least several of the representative populations comprising the ESU and the recovery population level. Performance standards were expressed as probabilities of being above the threshold escapement levels over 24 or 100 years (referred to as “24-year survival standard” and “100-year survival standard”) and reaching an average escapement equal or greater than the recovery level in years 41-48 (referred to as the “48-year recovery standard”). Actions were considered to have acceptable probabilities of meeting these standards if there is at least a 70% probability of meeting the 24- and 100-year survival standards and at least a 50% probability of meeting the 48-year recovery standard.

There are several reasons why applying the estimates of population levels for Snake River chinook to Snake River steelhead are problematic at this time. For example, the “threshold” levels for Snake River chinook were defined at least partially by the levels at which population simulation model behavior was uncertain and such simulation models currently do not exist for Snake River steelhead. Also, recovery population levels associated with delisting Snake River chinook were developed by the Snake River Recovery Team and NMFS following a multi-year process. Recovery population levels for Snake River steelhead have not yet been defined[22] and are unlikely to be defined until NMFS develops proposed delisting criteria, which may take several years.

This report acknowledges the limitations of steelhead data and modeling tools and proposes a more qualitative approach to looking at effects of proposed management actions on steelhead, based on inferences from the more detailed PATH spring/summer chinook analysis. This report draws largely from information and techniques included in the National Marine Fisheries Service (NMFS) 1998 Supplemental Biological Opinion, which reviewed effects of operation of the Federal Columbia River

22 If a recovery escapement level for the aggregate Snake River steelhead ESU were to be defined in a manner consistent with the recovery level for Snake River spring/summer chinook (60% of mean pre-1971 escapements), the 8-year geometric mean escapement past Lower Granite Dam would have to equal 37,568 wild adults.

Power System (FCRPS) on steelhead. The approach in that document was closely coordinated with the PATH steelhead work group.

4.2 Overview of Method Used to Infer Performance of Management Options Relative to Snake River Steelhead from Snake River Spring/Summer Chinook Model Results

Because the analytical method used for spring/summer chinook is not possible to apply at this time, an alternative method that draws inferences from the spring/summer chinook analysis is proposed. This method is outlined below as sequential steps and then each step is discussed in greater detail in the remainder of the report.

1. Determine whether spring/summer chinook management actions that result in an acceptable probability of being above survival threshold levels and reaching recovery levels correspond to historical smolt-to-adult survival rates (SAR). Assume that, if this correspondence exists for Snake River spring/summer chinook salmon, then it will also exist for Snake River steelhead.

2. Define a historical range of Snake River steelhead smolt-to-adult survival rates (SAR) as a proxy for an acceptable probability of being above survival threshold levels and reaching recovery levels.

3. Define the incremental change from recent steelhead SARs that is necessary to achieve historical SARs.

4. Compare the steelhead incremental survival change with a similar increment estimated for Snake River spring/summer chinook salmon.

5. Determine if the management action is likely to have a similar effect on Snake River steelhead hydrosystem survival, compared to Snake River spring/summer chinook hydrosystem survival.

6. Determine if the management action is likely to have a similar effect on Snake River steelhead survival outside the hydrosystem, compared to Snake River spring/summer chinook survival outside the hydrosystem.

7. Assume that ***if***:

    a) spring/summer chinook management actions that result in an acceptable probability of being above survival threshold levels and reaching recovery levels correspond to historical smolt-to-adult survival rates (SAR);

    then historical SARs can be used as a proxy for an acceptable probability of survival and recovery in Snake River spring/summer chinook ***and*** that this approach can be extended to Snake River steelhead.

    Assume further that ***if***:

    b) the incremental change between current and historical SAR for steelhead is less than or equal to the incremental change for spring/summer chinook;

    c) the management action is likely to have a similar effect on both Snake River steelhead and spring/summer chinook hydrosystem survival;

d) the management action is likely to have a similar effect on both Snake River steelhead and spring/summer chinook hydrosystem survival outside of the hydrosystem; and
e) a management action results in an acceptable probability of Snake River spring/summer chinook meeting survival and recovery goals;

then it is likely that the management action will result in an acceptable probability of survival and recovery for Snake River steelhead.

If a proposed management action results in an acceptable likelihood of survival and recovery for spring/summer chinook, this method evaluates whether or not it also results in an acceptable likelihood for steelhead. However, if an action fails to result in an acceptable likelihood of survival and recovery for spring/summer chinook, this method does not address whether steelhead have an acceptable or unacceptable likelihood. If, in Step 4, it is determined that the incremental survival change needed for steelhead is less than that needed for chinook, it is possible that some actions that are unacceptable for chinook may be acceptable for steelhead.

4.3 Detailed Evaluation of Point #1: Spring/Summer Chinook Management Actions that Result in an Acceptable Probability of Being Above Survival Threshold Levels and Reaching Recovery Levels Correspond to Historical Smolt-to-Adult Survival Rates (SAR)

Smolt-to-adult return rates (SAR) are a way of using readily available data to address what is conceptually a very simple question. Given a specific number of smolts migrating from a particular tributary at low seeding levels, what fraction of that number of smolts must return as adults to produce a greater number of smolts outmigrating in the next generation? Once the run is rebuilt, smolt-to-adult survivals at this rate would be expected to sustain the run at healthy levels. This survival rate is estimated as a range to reflect year-to-year variations in environmental conditions. It is important to remember that survival through the smolt-to-adult life stages incorporates a combination of human-induced mortality from several sources (e.g., dam passage, water management, harvest, estuarine/riverine habitat modifications, and water quality modifications) as well as natural mortality associated with conditions in the river, estuary, and ocean. The SAR does not explicitly incorporate survival throughout the entire life cycle, as does the simulation modeling conducted by PATH for Snake River spring/summer chinook salmon. That is, the SAR analysis does not incorporate the survival rate of adults from the time they pass above the upper dam until spawning or the survival of their progeny until these reach the upper dam as smolts. Therefore, the range of SARs must be chosen to provide a high enough rate of survival through the smolt-to-adult life stages that overall survival throughout the life cycle will be adequate for the species to persist and recover.

The only currently available method of identifying the range of SARs adequate for survival and recovery of an ESU is to look at the SARs that were achieved historically, when the ESU was at recovery population levels and survival rates were not declining. The PATH analytical group has suggested a correspondence between the SARs of aggregate Snake River spring/summer chinook stocks during the 1960s, when these stocks appeared to be “healthy” (i.e., experiencing high survival, with populations at or near the proposed recovery levels), and the ability of the stocks to persist and recover (Chapter 6 in Marmorek et al. 1996). A PATH subcommittee suggested an interim SAR goal of two to six percent until more quantitative analyses of stock performance could be completed. The PATH interim SAR goal was derived from consideration of three different approaches: 1) estimates of Snake River spring/summer chinook SARs to the upper dam during a relevant historical period; 2) back-calculation of theoretical

SARs from Snake River spawner-to-smolt survival estimates; and 3) comparison with SARs from a lower Columbia River stock (Warm Springs). The first method has the greatest potential for application to other species, because, for example, spawner-to-smolt survival estimates are not available for most steelhead stocks and steelhead SARs do not exist for the Warm Springs stock. This was implemented in Chapter 6 in Marmorek et al. (1996) by considering both Raymond's (1988) estimates, which were expressed as escapement and harvest [(SAR to upper dam)/(1-Harvest)] for wild Snake River spring and summer chinook stocks during the 1960s, and also by estimating SAR as escapement to the upper dam. This was done by multiplying Raymond's (1988) estimates by (1 - average harvest rates during the 1960s). This calculation, using approximate harvest rates, yielded SARs to the upper-most dam ranging from two to four percent for Snake River spring chinook and from two to five percent for Snake River summer chinook (Chapter 6 in Marmorek et al. 1996). Raymond's (1988) method resulted in estimates ranging from approximately three to six percent during the same period.

Some recent work by PATH supports the idea that historical SARs are more than adequate for meeting the 100-year survival and 48-year recovery goals for Snake River spring/summer chinook salmon, but may be inadequate for meeting the 24-year survival goal (Figure 4.3-1). This analysis is based on SARs (expressed as escapement to the upper dam) generated by the PATH life-cycle model, as described on p. 32 of Marmorek and Peters (1998a).

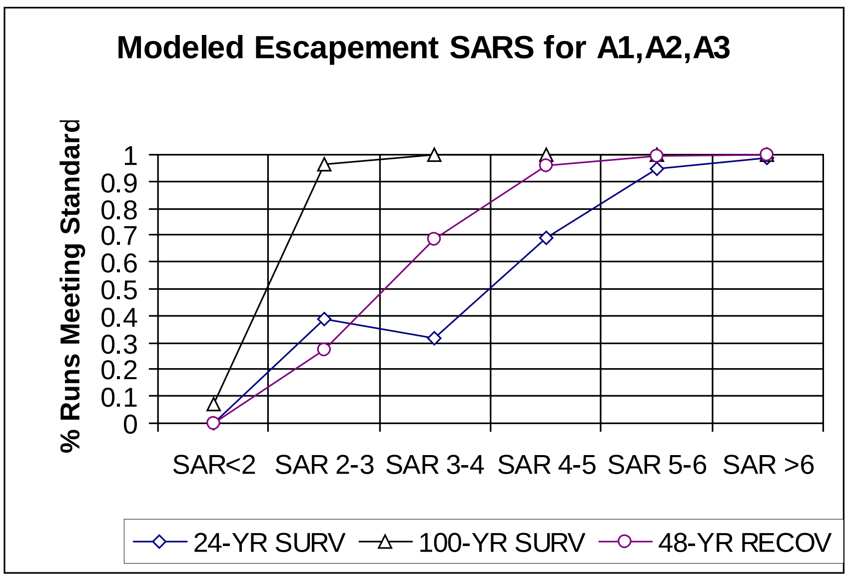

Figure 4.3-1: Probability that model runs resulting in 100-year median escapement SAR (generated by PATH life-cycle model as SAR to the upper dam) meet jeopardy standards for Snake River spring/summer chinook salmon. For example, for model runs resulting in a simulated median escapement SAR between 3.0 and 3.99, slightly more than 30% of these runs meet the 24-year survival standard, slightly less than 70% meet the 48-year recovery standard, and all of them meet the 100-year survival standard. Certainty of meeting the 100-year survival standard requires a median escapement SAR of at least 3%, certainty of meeting the 48-year recovery standard requires a median escapement SAR of at least 4%, and certainty of meeting the 24-year survival standard requires a median escapement SAR greater than 6%.

4.4 Detailed Evaluation of Point #2: Historical Range of Snake River Steelhead Smolt-to-Adult Survival Rates (SAR), for Use as a Proxy for an Acceptable Probability of Being Above Survival Threshold Levels and Reaching Recovery Levels

Four key questions were identified in development of this approach.

First, what is the specific historical period that was associated with a "healthy population?" Examination of Figure 4 in Raymond (1988) and Figure 1 in Petrosky and Schaller (1998) suggests that, while some estimates indicate that wild Snake River steelhead survival may have begun to decline prior to the 1970 smolt outmigration, a declining trend had clearly begun by 1970. The PATH subcommittee considered it reasonable to define the relevant historical period as 1964 through 1969, the same as that defined for Snake River spring/summer chinook salmon (see below), although the choice of this specific time period was less obvious for Snake River steelhead.

Second, what is the appropriate definition of SAR? The two historical SAR definitions described in Chapter 6 in Marmorek et al. (1996) were considered by the PATH subcommittee, but consensus has not yet been reached regarding the most appropriate definition to use as a surrogate for an acceptable probability of survival (spawning escapement above threshold escapement levels) and recovery. Therefore, both are applied in this report. It does not appear that differences between the two definitions result in different conclusions in the application of this analytical approach, assuming that both are applied consistently to the historical and recent time periods as described below. Briefly, the two definitions are as follows.

Smolt-to-adult return rate to the upper dam ("Escapement SAR").
This first definition is simply the number of adults from a given outmigration year that returned to the most upstream dam, divided by the number of smolts that passed the same dam during the outmigration year. This SAR definition is estimated for each year during the historical period when escapement and survival rates were considered adequate for species persistence and recovery. This SAR definition does not directly assume a particular level of harvest, although the harvest rates that occurred during the historical period affected the number of adults surviving to the upper dam. Various combinations of harvest and other sources of human-induced mortality could be combined to attempt to match historical SARs under this definition. This definition simply captures the overall survival rate through the FCRPS, lower river, and ocean that appears to be associated with an adequate level of historical survival and escapement.

Smolt-to-adult return rate to the upper dam, adjusted for harvest ("Escapement + Harvest SAR").
This definition differs from the previous definition by adding the number of harvested adults to the number of adults returning to the upper dam. A simple way of relating this definition to the first is to express it as the SAR to the upper dam (Escapement SAR), divided by (1 - the harvest rate). Because historical harvest rates were relatively high, estimates resulting from this "Escapement + Harvest SAR" definition are higher than estimates resulting from the "Escapement SAR" definition. These Escapement + Harvest SAR survival rates represent the "potential survival" of the population to the upper dam, if harvest had not occurred during the historical period. Escapement + Harvest SAR may also be thought of as representing survival to the mouth of the Columbia River during the historical period, reduced by mortality from sources other than harvest between the river mouth and the upper dam.

Third, what combination of stocks should be included in the SAR estimation for a given ESU? Ideally, there should be a separate estimate of SAR for each Snake River steelhead population, but this information does not exist. Some PATH members believe that Snake River steelhead "A-run" and "B-run" components should be separated in SAR analyses. However, 1) this information is not available at this time (Petrosky 1998a); 2) there is some confusion about the exact definition and the biological significance of "A-run" and "B-run" designations (Busby et al. 1996); and 3) it is not clear what grouping of Snake River spring/summer chinook populations would be compared with these groupings of Snake River steelhead. The limitations of the information available at the present time dictate that Snake River steelhead SARs represent an aggregate for the entire ESU.

Fourth, what are the appropriate methods and sources of data to use in estimating historical SARs according to the above definitions? For this report, the general approach described in Petrosky (1998a) is applied. Preliminary estimates of Escapement SAR for Snake River steelhead during 1964 through 1969 range from 3.4% to 4.2% with a geometric mean of 3.8% (Table 4.4-1; Figure 4.4-1; Petrosky 1998a). Corresponding estimates for the Escapement + Harvest SAR definition range from 4.5% to 6.4% with a geometric mean of 5.6% (Table 4.4-2; Figure 4.4-2).

Table 4.4-1: Smolt-to-adult return rate (SAR) estimates to upper dam (Escapement SAR) during historical and recent periods for Snake River spring/summer chinook salmon and Snake River steelhead. (Petrosky 1998a; Petrosky and Schaller 1998).

	Snake River Spring/Summer Chinook	Snake River Steelhead
Historical SAR Range (Geometric Mean)	0.023 - 0.045 (0.029)	0.034 - 0.042 (0.038)
Recent SAR Range (Geometric Mean)	0.002 - 0.010 (0.004)	0.011 - 0.012 (0.011)
Necessary Incremental Change (Historical Mean ÷ Recent Mean)	6.9x	3.4x

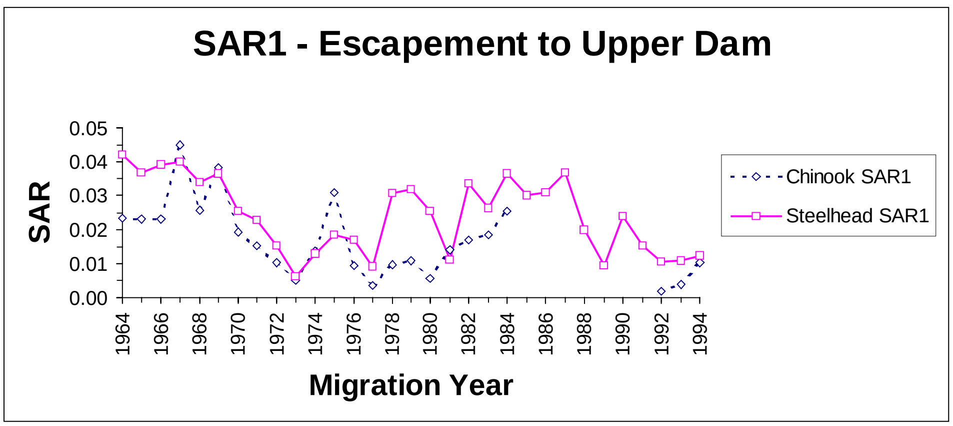

Figure 4.4-1: Estimates of Escapement SAR (to upper dam) for Snake River spring/summer chinook salmon (STH-SAR1) and Snake River steelhead (SCK-SAR1) from Petrosky (1998a) and Petrosky and Schaller (1998).

Table 4.4-2: Smolt-to-adult return rate (SAR) estimates to upper dam, adjusted for harvest (Escapement + Harvest SAR) during historical and recent periods for Snake River spring/summer chinook salmon and Snake River steelhead (Petrosky 1998a; Petrosky and Schaller 1998).

	Snake River Spring/Summer Chinook	**Snake River Steelhead**
Historical SAR Range (Geometric Mean)	0.037- 0.073 (0.049)	0.045 - 0.064 (0.056)
Recent SAR Range (Geometric Mean)	0.002 - 0.011 (0.004)	0.013 - 0.015 (0.013)
Necessary Incremental Change (Historical Mean ÷ Recent Mean)	11.2x	4.1x

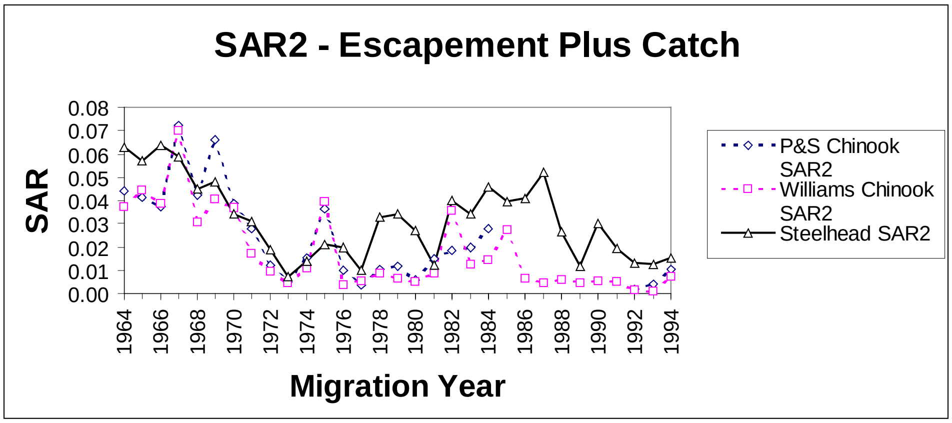

Figure 4.4-2: Estimates of Escapement + Harvest SAR for Snake River spring/summer chinook salmon (STH-SAR2) and Snake River steelhead (SCK-SAR2) from Petrosky (1998a) and Petrosky and Schaller (1998).

These SAR estimates are subject to change as PATH investigates alternate sources of data, assumptions, and methods of estimating the SARs of steelhead and spring/summer chinook salmon. PATH urges caution in interpreting these SAR estimates as absolute survival targets. Several concerns regarding the methods and assumptions employed in both Raymond (1988) and Petrosky and Schaller (1998) have been articulated by Paulsen and Giorgi (1998) and Williams (1998). Of particular concern are changes in the field sampling and analytical methods between historical and recent periods (Paulsen and Giorgi 1998), the significance of which are poorly understood at present. Some of these concerns can be addressed through sensitivity analyses (e.g., Petrosky 1998b) and some have been disputed by other PATH members (Petrosky et al. 1998). In short, several issues require further evaluation and discussion before a final PATH recommendation regarding use of any specific SAR as a proxy for survival and recovery standards can be developed. However, because most of the concerns, especially those relating to inconsistencies between historical and recent SAR estimates, appear to have similar effects on both steelhead and chinook SARs, it is unlikely that further PATH discussions will invalidate the use of these SARs for the ***relative*** comparisons between the two species described in this report.

4.5 Detailed Evaluation of Point #3: Incremental Change from Recent Steelhead SARs That is Necessary to Achieve Historical SAR

The preliminary estimate of the necessary historical Escapement SAR range for Snake River steelhead is 3.4% to 4.2% (geometric mean 3.8%). Escapement SARs for the three most recent available smolt migration years (1992-1994), estimated by similar methods, range from 1.1% to 1.2% (geometric mean 1.1%), and the mean incremental survival change required to meet historical SARs is a factor of 3.4 times the recent survival rates (Table 4.4-1; Petrosky 1998a). Using the Escapement + Harvest SAR definition, corresponding estimates are a historical mean of 5.6%, a recent mean of 1.3%, and a necessary 4.1-fold increase in survival over recent rates (Table 4.4-2; Petrosky 1998a).

4.6 Detailed Evaluation of Point #4: Comparison of Snake River Steelhead and Spring/Summer Chinook Incremental Changes

The tentative conclusion of the working group, based upon an inspection of trends in SARs and escapement, is that the period prior to the 1970 smolt outmigration is most closely associated with healthy population levels of wild Snake River spring/summer chinook salmon. This period, which preceded a sharp decline in both wild spring/summer chinook survival and spawner escapement, began with, or followed shortly after, the 1970 outmigration (see Figure 4 in Raymond 1988). The earliest smolt outmigration year for which historical SARs are available is 1964. Therefore, the historical time period corresponding to the necessary SAR is defined as 1964 through 1969.

Ideally, there should be a separate estimate of SAR for each Snake River spring/summer chinook index population that would correspond with the estimates of survival and recovery probabilities for the index population called for by both the BRWG (1994) and the jeopardy standard in the 1995 FCRPS Biological Opinion. However, this information was not available either for Snake River spring/summer chinook salmon or for the Snake River steelhead ESU that would be directly comparable to Snake River spring/summer chinook. Various methods for partially disaggregating this and other ESUs were discussed within the PATH subcommittee. For example, Raymond (1988) reported spring and summer components of the ESU separately, but this information is not currently available for recent years (Petrosky and Schaller 1998). Although population-level indicators of the species-level biological requirements of the Snake River spring/summer chinook ESU would be desirable, the best available tool was an aggregate SAR estimate for the entire ESU.

Petrosky and Schaller (1998) have produced preliminary estimates of both Escapement SAR and Escapement + Harvest SAR of Snake River spring/summer chinook based upon Raymond's (1988) original estimates, historical harvest, and estimates of the historical age structure of naturally spawning Snake River index stocks. Williams (1998; Williams et al. 1998) produced alternative estimates of spring/summer chinook Escapement + Harvest SAR, which are based upon an alternative historical age structure. The Williams (1998) estimates are nearly identical in all years, except 1969 and 1982-84 (Figure 4.1-3). The Petrosky and Schaller (1998) spring/summer chinook estimates are used in the remainder of this report because they are most comparable with the methodology used to generate the steelhead SAR estimates (Petrosky 1998) and because the two methods appear to differ only slightly for the years encompassing the "historical" and "recent" periods.

Preliminary estimates of the 1964 through 1969 Escapement SAR of Snake River spring/summer chinook range from 2.3% to 4.5% with a geometric mean of 2.9% (Table 4.4-1; Petrosky and Schaller 1998). Corresponding estimates of Escapement + Harvest SAR range from 3.7% to 7.3% with a geometric mean of 4.9% (Table 4.4-2). Escapement SARs for the three most recent available smolt migration years (1992-

1994), estimated by similar methods, range from 0.2% to 1.0% (geometric mean 0.4%), and the mean incremental change in survival required to meet historical Escapement SARs is a factor of 6.9 times the recent survival rates (Table 4.4-1; Petrosky and Schaller 1998). Using the Escapement + Harvest SAR, corresponding estimates are a historical mean of 4.9%, a recent mean of 0.4%, and a necessary 11.2-fold increase in survival over recent rates (Table 4.4-2; Petrosky and Schaller 1998).

The sensitivity of these estimates to alternative assumptions is currently being reviewed by a PATH subcommittee and estimates are subject to change prior to completion of the final PATH report. Of particular concern are changes in the field sampling and analytical methods between historical and recent periods (Paulsen and Giorgi 1998), the significance of which are poorly understood at present.

Both of the approaches summarized in Tables 4.4-1 and 4.4-2 indicate that the incremental smolt-to-adult survival change necessary to bring recent survival levels up to the mean historical level is less for Snake River steelhead than for Snake River spring/summer chinook salmon. This suggests that a set of management actions that results in an adequate likelihood of survival and recovery for Snake River spring/summer chinook is likely to do the same for Snake River steelhead if the action has a similar incremental effect on survival of each species.

4.7 Detailed Evaluation of Point #5: Action Will Have Similar Hydrosystem Survival Effect on Steelhead and Chinook

Two approaches are applied to answering this question. The first compares the routing and survival of steelhead and spring/summer chinook through the hydropower system to demonstrate that operation of the system in its current configuration appears to have similar effects on juveniles and adults of each species. The second approach compares the specific management actions considered by PATH to date, with respect to possible differential effects on direct passage survival of steelhead and spring/summer chinook.

4.7.1 Comparison of Steelhead vs Spring/Summer Chinook Routing Factors and Route-Specific Survivals Through the Hydrosystem, as Currently Operated and Configured

Juvenile Reservoir Survival

Reservoir-specific estimates of survival are not available for either juvenile chinook salmon or steelhead. Reach survival estimates, which include effects of dams in addition to effects of water regulation and impounded reservoirs, are described below in the section on *In-River Juvenile Survival Under Current Configuration and Operations*

Juvenile Migration Speed

Migration speed of steelhead at a given flow is generally greater in steelhead than in yearling chinook salmon, suggesting that exposure time to predation may be lower in steelhead and estuary arrival timing may be closer to that under which the species evolved (Table 4.7.1-1). Migration speed is positively correlated with flow for Snake River steelhead and Upper Columbia River steelhead, as reviewed in NMFS (1998).

Table 4.7.1-1: Comparison of migration speed of yearling chinook and steelhead in the Snake and Columbia Rivers, from bivariate flow:travel time relationships. Flows of 85 to 100 kcfs are examined because these correspond to the spring Snake River flow objectives to which Snake River chinook and steelhead are exposed. Closest flows to 85 to 100 are examined in the mid-Columbia to indicate similar flows for Upper Columbia steelhead.

Flow and Location	Migration Speed of Yearling Chinook	Migration Speed of Steelhead	Study
85-100 kcfs Lower Granite Dam to McNary Dam	11.6-12.7 mi/day (hatchery)	13.6-16.2 mi/day (hatchery)	Berggren and Filardo (1993)
85-100 kcfs Lower Granite Dam to Lower Monumental Dam	7.2-7.6 mi/day (hatchery)	7.4-9.2 mi/day (hatchery)	Smith et al. (1998) - SH range includes annual variability 1994-96
85-100; 110-130 kcfs Rock Island Dam to McNary Dam	11.0-11.8 mi/day at 85-100; 12.3-13.2 mi/day at 110-130 (Mixed)	14.2-16.0 mi/day at 85-100; 17.2-19.6 mi/day at 110-130 (Wild)	Giorgi et al. (1997)
85-100; 110-130 kcfs Methow River to McNary Dam	9.0-9.1 mi/day at 85-100; 9.1 mi/day at 110-130 (Hatchery)	12.0-13.8 mi/day at 85-100; 13.7-14.8 mi/day at 110-130 (Hatchery)	Berggren and Filardo (1993)

Flow versus Juvenile Survival

Correlative relationships between flow and juvenile reach survival are similar for both Snake River steelhead and Snake River spring/summer chinook salmon. A relationship between flow and reach survival within years was not detected for either species (Smith et al. 1998). However, relationships between mean annual flow and mean survival have been detected. Regression of reach survival of primary release groups between Lower Granite Reservoir and Lower Monumental Dam for 1994 through 1996 on flow was highly significant for each species, with comparable predictive power (R^2 = 0.65 for yearling chinook and R^2 = 0.52 for steelhead; Smith et al. 1998). The slopes of the regression lines also were nearly identical (0.0040 for chinook and 0.0038 for steelhead), suggesting a similar association between mean annual flow and mean annual survival in the two species. Regressions of the survival of daily release groups between Lower Granite Dam and Lower Monumental Dam for 1994 through 1996 against flow also were highly significant and very similar for the two species, although the predictive power of the regressions was poor (R^2 = 0.18 for chinook and R^2 = 0.17 for steelhead).

Correlative relationships between flow and smolt-to-adult survival are also similar for both Snake River steelhead and Snake River spring/summer chinook salmon. A significant relationship between SAR and average flow during juvenile Snake River spring/summer chinook salmon emigration was reviewed in NMFS (1995). Similarly, Petrosky (1998a) reported significant negative relationship between SAR for wild spring/summer chinook and steelhead and Lewiston-to-Bonneville water travel time for the 1964-1994 smolt migrations ($P < 0.001$ and < 0.001; and r^2 = 0.53 and 0.36, respectively). Water travel time is a function of the number and volume of reservoirs, and flow. Water travel time during 1964-1969 with 4-5 dams ranged from 5 to 17 days, depending on number of reservoirs and flow. In contrast, water travel time with 8 dams (1975-1994) ranged from 13 to 39 days, depending on flow.

Interpretation of both the reach survival and SAR correlations with flow, relative to inferring the efficacy of management actions such as flow augmentation and lowering reservoir elevations, is a matter of some debate. Rationale supporting application of these correlations to management actions is included in

NMFS (1998) and Petrosky (1998a). Caveats regarding this application are included in NMFS (1998). Conclusions regarding this application should be nearly identical for both steelhead and spring/summer chinook.

Juvenile Reservoir Predation Mortality

Limited information on predation rates in John Day Reservoir suggests that predators such as squawfish and smallmouth bass cause a higher mortality rate for juvenile steelhead (mixture of Snake River, Upper Columbia River, and Middle Columbia River steelhead) than for juvenile yearling chinook (Table 4.7.1-2). If the observations in John Day Reservoir are applicable to other reservoirs, removal of a given number of squawfish should reduce the steelhead predation rate more than the yearling chinook predation rate. (For example, Table 4.7.1-2 suggests that if half of the predators could be removed in April, steelhead mortality in John Day Reservoir for that month would presumably be reduced 6% while yearling chinook mortality would be reduced by 4%).

Table 4.7.1-2: Estimated mortality of juvenile salmon and steelhead from predation by month in John Day Reservoir, 1983 through 1986. Predators considered were northern squawfish (*Ptychocheilus oregonensis*), walleye (*Stizostedion vitreum*), and smallmouth bass (*Micropterus dolomieu*). During April and May virtually all chinook salmon are yearling chinook salmon, based on McNary Dam smolt monitoring. During July, chinook salmon are a mixture of yearlings and subyearlings, so the reported mortality rate is greater than the yearling chinook mortality rate. (From Table 6 in Reiman et al. 1991).

Month	Steelhead		Chinook Salmon	
	Mortality Rate	**Standard Deviation**	**Mortality Rate**	**Standard Deviation**
April	0.12	0.061	0.08	0.034
May	0.11	0.031	0.11	0.017
June	0.13	0.089	0.07	0.025

Juvenile Turbine Survival

Turbine survival studies published through 1990 at Snake and lower Columbia River dams have been reviewed by Iwamoto and Williams (1993). The Independent Scientific Group (ISG 1996) and Whitney et al. (1997) reviewed studies published through 1995, including several from mid-Columbia projects. At least one other turbine survival study has been conducted since that time (Normandeau Associates and Skalski 1997).

Turbine mortality has been estimated primarily for juvenile salmon, although at least three studies have estimated steelhead mortality (Weitkamp et al. 1980; Olson and Kaczynski 1980; Muir et al. 1998). Estimates of turbine mortality vary greatly among studies, ranging from 2.3% to 19%. Whitney et al. (1997) pointed out that studies that recovered marked fish in the tailrace very quickly using radio-tags resulted in estimates of seven percent mortality or less (average 5.5%). Results of Normandeau Associates and Skalski (1997), not reviewed by Whitney et al. (1997), fit into this category as well. According to Whitney et al. (1997), 18 other studies with longer times between turbine passage and recovery averaged 10.9% mortality. The same report suggested that the lower estimates most likely estimate mortality directly associated with turbine passage while the others probably include factors beyond the turbine, such as predation of disoriented smolts.

With only three estimates of steelhead turbine mortality among the more than 20 estimates for salmon, it is not possible to determine whether survival rates differ between species. The range of estimates in the steelhead studies (3% to 16%) is similar to the range in chinook studies (2.3% to 19%), but may be more closely related to the type of turbine involved. The steelhead estimate for passage through Kaplan turbines, using a "long" recovery period technique, was 16%. This mortality rate is similar to the level observed in the majority of the "long" recovery chinook studies, most of which involved passage through Kaplan turbines. The steelhead mortality estimate for passage through bulb turbines was 3%, directly comparable to a mortality estimate of seven percent for coho salmon in the same study.

In summary, it is unlikely that turbine survival rates of steelhead are different from those for yearling chinook salmon.

Juvenile Spill Survival

Whitney et al. (1997) reviewed 13 estimates of spill mortality (3 steelhead and 10 salmon) published through 1995 and concluded that zero to two percent is the most likely range for standard spill bays. However, they also pointed out that local conditions, such as back eddies or other situations that may favor the presence of predators, may lead to higher spill mortality. In some studies reviewed by Whitney et al. (1997), point estimates for mortality in spill bays with spill deflectors were higher than estimates for spill bays without deflectors, but there were no statistical differences between the two. This also occurred in two more recent studies (Muir et al. 1997; Dawley et al. 1997), but there were significant differences between the two spillway types in another recent study (Normandeau et al. 1996) and a fourth study showed statistically significant differences at one flow rate but not at others (Mathur et al. 1997).

In general, steelhead spill survival estimates are the same as salmon spill survival estimates, since two of three available estimates are 0% to 2.2%. One exception is an estimate of 27.5% steelhead mortality associated with passage through a spill bay without a deflector at Lower Monumental Dam (Long et al. 1975). In the same study, the authors found a more normal spill mortality rate (2.2%) associated with a spillbay equipped with a deflector. The authors recognized that the "without-deflector" result was highly unusual and proposed that a condition favoring predation below the test spillway may have affected results.

Juvenile Bypass Survival

Direct bypass survival is defined as survival past systems including turbine intake screens, gatewells, orifices, bypass flumes, and, in some cases, dewatering screens, wet separators, sampling facilities including holding tanks, and bypass outfall conduits. Indirect bypass mortality may be associated with predation that occurs at a poorly-sited bypass outfall or delayed mortality caused by bypass passage, but expressed further downstream. A minimum estimate of mortality can be determined from observations of dead fish in sampling facilities. Table 4.7.1-3 summarizes recent yearling chinook and wild steelhead facility mortality estimates at juvenile sampling facilities in recent years. These estimates suggest that direct bypass mortality of both wild steelhead and yearling chinook is generally less than 1% and that in nearly all cases juvenile steelhead facility mortality is less than yearling chinook mortality.

No measure of indirect mortality (following outfall release) is available at most projects for juvenile steelhead. However, studies of subyearling chinook bypass mortality at Bonneville Powerhouse One and Powerhouse Two (Ledgerwood et al. 1990, 1994; Dawley et al. 1996) indicate that high bypass mortality may be associated with predation that occurs at a poorly-sited bypass outfall. There is no information to suggest that indirect mortality is higher for steelhead than for yearling chinook salmon at any projects under current conditions.

Table 4.7.1-3: Percent facility mortality at juvenile fish facilities, 1993 to 1996, from Martinson et al. (1997) and Spurgeon et al. (1997).

Dam and Year	Yearling Chinook (Mixed)	Yearling Chinook (Wild)	Steelhead (Wild)	Difference (SH - CH)
Lower Granite				
1993		0.003	<0.001	-0.002
1994		0.004	<0.001	-0.003
1995		0.002	<0.001	-0.001
1996		0.009	<0.001	-0.008
Little Goose				
1993		0.004	0.001	-0.003
1994		0.012	0.002	-0.010
1995		0.006	0.001	-0.005
1996		0.012	0.002	-0.010
Lower Monumental				
1993		0.001	0.001	0.000
1994		0.005	0.003	-0.002
1995		0.002	0.001	-0.001
1996		0.004	0.001	-0.003
Ice Harbor				
1996		0.000	0.000	0.000
McNary				
1993	0.006		0.002	-0.004
1994	0.011		0.005	-0.006
1995	0.001		<0.001	0.000
1996	0.001		0.001	0.000
Bonneville PH1				
1993	0.001		0.000	-0.001
1994	0.002		0.001	-0.001
1995	0.001		0.000	-0.001
1996	0.002		0.001	-0.001
Bonneville PH2				
1993	0.007		0.000	-0.007
1994	0.013		0.004	-0.009
1995	0.006		0.015	0.009
1996	0.005		0.005	0.000

Fish Guidance Efficiency

The effectiveness of intake screens in diverting fish approaching the turbines into bypass systems is known as fish guidance efficiency (FGE). FGE differs among wild and hatchery yearling chinook salmon

(Krasnow 1998) but appears to be identical for wild and hatchery steelhead (S. Smith, NMFS, pers. comm. 1998), based on analysis of recent PIT-tag detection rates. For both species, there is uncertainty regarding the change in FGE occurring since the replacement of standard length traveling screens (STS) with extended-length bar screens (ESBS) at several projects (Krasnow 1998; Marmorek and Peters 1998b). Side-by-side estimates of STS versus ESBS FGE using fyke-net recoveries indicate that FGE is considerably higher with ESBS than with STS (e.g., McComas et al. 1993; Gessel et al. 1994; Brege et al. 1994). However, this difference has not been confirmed under full operating conditions, based on PIT-tag detection rates before and after ESBS installation at Snake River projects (IDFG analysis reported in Krasnow 1998). The PATH analytical group has recommended examining sensitivity to both assumptions.

FGE estimates in Table 4.7.1-4 show that relative guidance of steelhead and yearling chinook salmon varies by project, chinook origin, and ESBS versus STS assumption. Steelhead FGE is estimated to be 3% to 29% higher than yearling chinook FGE at all projects except McNary, The Dalles (which does not have a screened bypass system), and the Bonneville second powerhouse.

Table 4.7.1-4: Estimates of fish guidance efficiency (FGE) with current project configurations, from Krasnow (1998) and Smith (1998). Estimates are made for two assumptions about the effectiveness of extended-length bar screens (ESBS), relative to standard traveling screens (STS) for chinook; however, these alternative assumptions do not apply to steelhead FGE estimates, based on available PIT-tag observations (Smith 1998). Position of operating gate affects FGE, as described in Krasnow (1998). ROG = raised operating gate; SOG = stored operating gate; LSTS = lowered standard traveling screen; STR = streamlined trash rack; TIE = turbine intake extension.

Dam	Current Fish Guidance Configuration/Structure	ESBS > STS				ESBS = STS				Difference (SH - Wild CH)
		Yearling Chinook			Steelhead	Yearling Chinook			Steelhead	
		Wild	Hatchery	Mixed		Wild	Hatchery	Mixed		
Lower Granite	6 of 6 turbines w/ ESBS, ROG	0.78	0.73	0.75	0.81	0.55	0.46	0.47	0.81	0.03 to 0.26
Little Goose	6 of 6 turbines w/ ESBS, ROG	0.82	0.76	0.78	0.81	0.64	0.52	0.54	0.81	0.01 to 0.17
Lower Monumental	6 of 6 turbines w/ STS, SOG	0.61	0.47	0.49	0.82	0.61	0.47	0.49	0.82	0.21
Ice Harbor	6 of 6 turbines w/ STS, ROG	0.71	0.60	0.62	0.74	0.71	0.60	0.62	0.74	0.03
McNary	14 of 14 turbines w/ ESBS, ROG	0.95	0.81	0.83	0.89	0.79	0.68	0.69	0.89	-0.06 to +0.10
John Day	16 of 16 turbines w/ STS, no OG	0.64	0.54	0.55	0.68	0.64	0.54	0.55	0.68	0.04
The Dalles	Ice and trash sluiceway w/ forebay overflow	0.46	0.39	0.40	0.40	0.46	0.39	0.40	0.40	-0.06
Bonneville I	10 of 10 units w/ STS, SOG	0.38	0.32	0.33	0.51	0.38	0.32	0.33	0.51	0.13
Bonneville II	8 of 8 units w/ LSTS, STR, alt TIE, SOG	0.44	0.37	0.38	0.39	0.44	0.37	0.38	0.39	-0.05

Spill Efficiency/Effectiveness

Spill effectiveness is the proportion of fish approaching a project that pass via the spillway, and spill efficiency is spill effectiveness divided by the proportion of total river flow that is passing over the spillway at the same time. Recent reviews of spill efficiency and effectiveness include Steig (1994), Giorgi (1996), Whitney et al. (1997), and Marmorek and Peters (1998b). Estimates of spill efficiency vary by project and, in some cases, can be described as functions of the proportion of project flow passing over the spillway. Nearly all spill efficiency studies are based on hydroacoustics, and therefore steelhead and yearling chinook efficiencies cannot be distinguished. One recent radio-telemetry study included relatively large numbers of each species, allowing seasonal-average comparisons of the proportion of fish that passed the dam by a known route (Adams et al. 1997 – their Figure 4-12). In this study, 37% of hatchery yearling chinook, 50% of hatchery steelhead, and 47% of wild steelhead went through the spillway over the course of the study period. The distributions of release-group timings for the two species were not identical, so the two species may not have been exposed to identical distributions of spill conditions. However, these results suggest that the seasonal effectiveness of spill in passing steelhead was at least as great as that observed for yearling chinook at Lower Granite Dam during 1996.

In-River Juvenile Survival Under Current Configuration and Operations

Recent (1994-1996) PIT-tag derived estimates of survival through the Lower Granite Reservoir to Lower Monumental Dam reach suggest that **wild** Snake River spring/summer chinook salmon survive at slightly higher rates (approximately 1.2% to 3.2% higher survival per project) than wild Snake River steelhead (Smith et al. 1998; Schiewe 1997; Table 4.7.1-5, Figure 4.4-1). Results of the 1994-1996 PIT-tag studies involving **hatchery fish or mixed hatchery and wild fish** through river reaches, which in some instances include estimates of survival to McNary Dam, are variable, with Snake River steelhead surviving at higher rates than Snake River spring/summer chinook in some cases and at lower rates in others. Results of earlier studies (1966-1977, 1980), which included only wild fish in the earliest years and mixed stocks in all others, are similar to the 1994-1996 mixed stocks results (Table 4.7.1-5, Figure 4.7.1-1). An exception is 1971, when steelhead per-project survival was approximately 20% higher than chinook survival.

Recent estimates of **wild** Snake River steelhead and chinook survival through projects below Lower Monumental Dam under recent conditions are not currently available. Ideally, passage-simulation models, such as those used for the PATH spring/summer chinook analysis, with alternative functional relationships suggested by the results of the Snake River survival study that are applied to lower river projects, could be used to estimate expected survival past all FCRPS projects under the proposed action. The two primary alternative passage simulation models in the region predicted dissimilar survival of Snake River spring/summer chinook salmon in recent PATH analyses (Marmorek and Peters 1998a) It is likely that the two models would also produce dissimilar results for steelhead. PATH considers it important to obtain results from both models when analyzing the effects of proposed actions in biological opinions. Results from one of the passage simulation models (CRiSP) were submitted to NMFS for use in the recent steelhead Supplemental Biological Opinion (Anderson 1998). The second passage simulation model (FLUSH) currently is not configured to generate steelhead survival estimates, so it is not possible to compare estimates generated by the two models. The CRiSP model results provided selected steelhead survival estimates for actions that approximate management action A1 (current configuration and operation of the FCRPS), but provided no corresponding spring/summer chinook survival estimates for comparison. These results are discussed anecdotally, but this report relies primarily upon a simpler approach to applying recent reach survival estimates to the entire hydrosystem migration corridor.

The recent PIT-tag reach survival estimates for wild smolts to Lower Monumental Dam in Table 4.7.1-5 indicate that the per-project survival difference between Snake River steelhead and Snake River spring/summer chinook salmon is negligible, as described above. However, the difference may be more significant if passage through all eight dams is considered. Additionally, the routing through dams of PIT-tagged fish in the passage survival studies differs from the passage routing of the majority of migrants, which were not PIT-tagged. This is because, during 1994 through 1996, all PIT-tagged fish that went into bypasses at Snake River transportation collector projects (Lower Granite, Little Goose, Lower Monumental, [and McNary for 1994 only]) were routed back to the river, but all un-tagged fish were collected for transportation. Therefore, inriver migrants that were not included in the survival studies passed Snake River collector projects only via the turbines or spillways and likely suffered higher mortality than did PIT-tagged experimental fish, which also could pass those projects via bypasses.

Table 4.7.1-5: Seasonal average estimates of survival of yearling chinook and steelhead released in the Snake River, ascertained from studies in 1966-1977, 1980, and 1994-1996 (Raymond 1979; Sims et al. 1977, 1978, 1981). A "project" refers to a dam + reservoir combination and a "reach" refers to river segment composed of one or more projects. Estimates in normal typeface are those reported in cited research. Bold estimates are calculated in this table as: Mean Per-Project Survival = (Reach Survival)$^{-(\text{Number of Projects})}$, Approximate Eight-Project Survival [from 1975 on] = (Mean Per-Project Survival)^{8}.

Study	Reach	Number of Projects	Steelhead Reach Survival (and Mean Per-Project Survival)	Approximate Eight-Project Steelhead Survival	Yearling Chinook Reach Survival (and Mean Per-Project Survival)	Approximate Eight-Project Yearling Chinook Survival	Difference in Per-Project (and Eight-Project) Survival [SH - CH]
1966	IHR-TDA	3	0.75 (0.91) (Wild)		0.63 (0.86) (W)		
1967	IHR-TDA	3	0.57 (0.83) (Wild)		0.64 (0.86) (W)		
1968	IHR-TDA	3	0.60 (0.84) (Wild)		0.62 (0.85) (W)		
1969	LMN-TDA	4	0.36 (0.77) (Wild)		0.47 (0.83) (W)		
1970	LGS-TDA	5	0.38 (0.82) (Mixed)		0.22 (0.74) (M)		
1971	LGS-IHR	2	0.80 (0.89) (Mixed)		0.48 (0.69) (M)		
1972	LGS-TDA	5	0.20 (0.72) (Mixed)		0.16 (0.69) (M)		
1973	LGS-TDA	5	0.04 (0.53) (Mixed)		0.05 (0.55) (M)		
1974	LGS-TDA	5	0.20 (0.72) (Mixed)		0.36 (0.82) (M)		
1975	LGR-TDA	6	0.41 (0.86) (Mixed)	0.30 (M)	0.25 (0.79) (M)	0.15 (M)	0.15 (M)
1976	LGR-JDA	5	0.36 (0.82) (Mixed)	0.20 (M)	0.30 (0.79) (M)	0.15 (M)	0.05 (M)
1977	LGR-JDA	5	0.02 (0.46) (Mixed)	0.002 (M)	0.03 (0.50) (Mixed)	0.003 (M)	-0.001 (M)
1980	LGR-JDA	5	0.21 (0.73) (Mixed)	0.08 (M)	0.36 (0.82) (M)	0.20 (M)	-0.12 (M)
1994 Primary Release Groups Weighted Means (Smith et al. 1998, Tables E1 and E2)	Silcott Island (LGR Reservoir) Lower Monumental Dam	3	0.590 (0.838) (Hatchery) [4/23-5/16]	0.243 (H)	0.645 (0.864) (Hatchery) [4/16-5/11]	0.311 (H)	-0.026 (-0.068) (H)
1994 Snake Trap Releases Weighted Means (Smith et al. 1998, Table E8)	Snake Trap (Above LGR Reservoir) - Lower Monumental Dam	3	0.351 (0.705) (Hatchery) [4/13-7/8] 0.515 (0.802) (Wild) [4/13-7/5]	0.061 (H) 0.171 (W)	0.571 (0.823) (Hatchery) [4/13-7/6] 0.580 (0.834) (Wild) [4/13-7/6]	0.210 (H) 0.234 (W)	-0.118 (-0.149) (H) -0.032 (-0.063) (W)
1995 Primary Release Groups Weighted Means (Smith et al. 1998, Tables E1 and E2)	Port of Wilma (LGR Reservoir) - Lower Monumental Dam	3	0.788 (0.924) (Hatchery) [4/22-5/12]	0.531 (H)	0.779 (0.920) (Hatchery) [4/9-5/5]	0.513 (H)	0.004 (0.018) (H)
1995 Snake Trap Releases Weighted Means (Smith et al. 1998, Table E9)	Snake Trap (Above LGR Reservoir) - Lower Monumental Dam	3	0.752 (0.909) (Hatchery) [3/31-5/31] 0.790 (0.924) (Wild) [3/31-5/31]	0.466 (H) 0.531 (W)	0.729 (0.900) (Hatchery) [3/31-5/31] 0.844 (0.945) (Wild) [3/31-5/31]	0.430 (H) 0.636 (W)	0.009 (0.036) (H) -0.021 (-0.105) (W)

Study	Reach	Number of Projects	Steelhead Reach Survival (and Mean Per-Project Survival)	Approximate Eight-Project Steelhead Survival	Yearling Chinook Reach Survival (and Mean Per-Project Survival)	Approximate Eight-Project Yearling Chinook Survival	Difference in Per-Project (and Eight-Project) Survival [SH - CH]
1995 Weekly Transport + Other Release Groups Unweighted Means (S. Smith, pers. comm. 1998)	Lower Granite Dam - McNary Dam	4.5	0.592 (0.890) (Mixed) [4/9-5/27]	0.394 (M)	0.624 (0.901) (Mixed) [4/4-6/12]	0.434 (M)	-0.011 (-0.040) (M)
1996 Weekly Transport + Other Release Groups Unweighted Means (S. Smith, pers. comm. 1998)	Lower Granite Dam - McNary Dam	4.5	0.615 (0.899) (Mixed) [4/11-5/29]	0.427 (M)	0.587 (0.888) (Mixed) [4/16-5/27]	0.387 (M)	0.011 (0.040) (M)
1996 Snake Trap Releases Weighted Means (Smith et al. 1998, Table 24)	Snake Trap (Above LGR Reservoir) - Lower Monumental Dam	3	0.954 (0.984) (Hatchery) [4/15-5/15] 0.951 (0.983) (Wild) [4/15-5/15]	0.879 (H) 0.872 (W)	0.703 (0.889) (Hatchery) [4/8-5/15] 0.963 (0.988) (Wild) [4/5-5/15]	0.390 (H) 0.908 (W)	0.095 (0.489) (H) -0.012 (-0.036) (W)
1997 Preliminary Information (Schiewe 1997b)	Lower Granite Dam - McNary Dam	4.5	0.640 (0.906) (Hatchery)	0.454 (H)	0.672 (0.915) (Mixed)	0.491 (M)	-0.009 (0.037) (M)

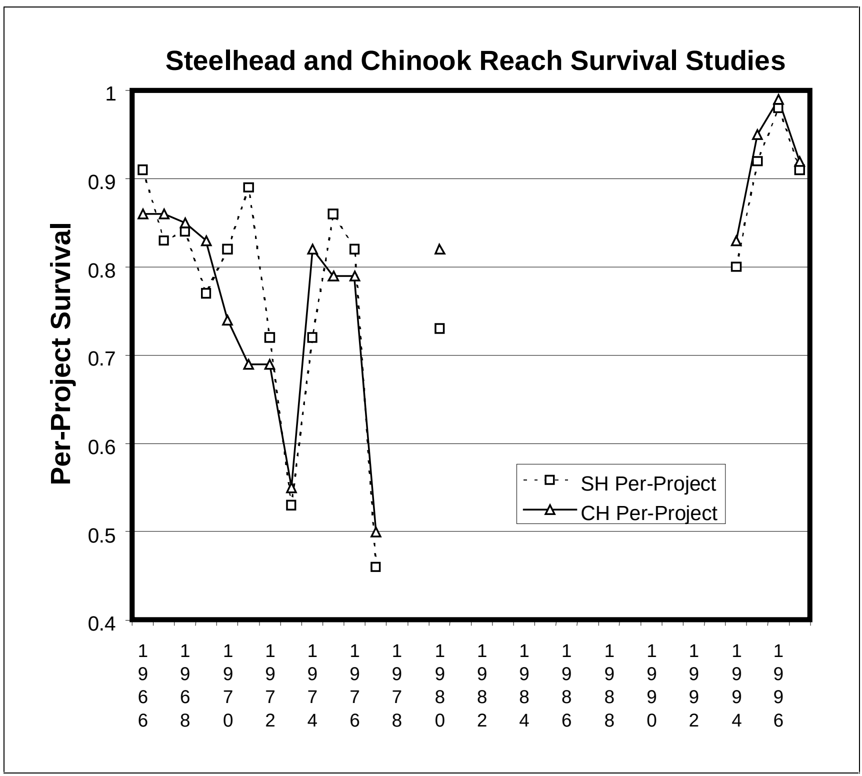

Figure 4.7.1-1: Snake River steelhead and Snake River spring/summer chinook salmon per-project reach survival estimates, from Table 4.7.1-5. Only wild steelhead and chinook estimates are presented for 1994-1996. All other years include a mixture of wild and hatchery stocks.

In the absence of inriver survival estimates from either passage model (Anderson 1998 estimated survival of combined inriver and transported fish, as discussed below), a very simple approach was used to estimate the relative survival of the two species through all eight dams. These estimates represent approximate survival of the two species through eight projects, estimated in an identical manner for each species, based on conditions experienced in 1994 through 1996. The primary assumptions for this simple approach are: 1) the survival of wild Snake River steelhead and Snake River spring/summer chinook through each project below Lower Monumental Dam are equal to the mean per-project survival through Lower Monumental Dam, as described in Table 4.7.1-5; and 2) survival of inriver migrants under the current operation (which transports all fish bypassed at three Snake River collector projects, leaving inriver fish to pass only through turbines or spill at those projects) are approximately 80% of the survival of fish passing inriver (pers. comm., P. Wilson, CBFWA [FLUSH model], and J. Hayes, Univ. of Washington [CRiSP model], 1998). The first assumption appears reasonable given estimates of survival for hatchery and mixed smolts through McNary Dam in Table 4.7.1-5 and given one estimate of mixed hatchery and wild spring/summer chinook survival through John Day Reservoir in 1996, which was similar to mean survival through Snake River projects (Smith et al. 1998). The second assumption is based on detailed routing estimates of spring/summer chinook in passage simulation model analyses included in Marmorek and Peters (1998b). Its application to steelhead is, at this point, speculative. Using this approach, the range of inriver survival differences between Snake River steelhead and Snake River spring/summer chinook through eight projects is approximately 3.6% to 10.5% (Table 4.7.1-5).

Direct Transportation Survival

PATH has estimated that direct survival of yearling chinook salmon during transportation is high, and an estimate of 98% has been used in modeling (Marmorek and Peters 1998b). There are no studies in which direct transportation survival of steelhead has been empirically estimated, but it is likely that 98% is also a reasonable estimate for steelhead direct transport survival.

Combined Transport and In-River Direct Survival Under Current Operations

Comparative passage model estimates of combined transported and inriver migrant Snake River steelhead and chinook survival to below Bonneville Dam are not currently available. A CRiSP-model estimate of a “spread the risk” action, presumably comparable to the current operation, given the 1998 predicted flows, indicated 71% survival of Snake River steelhead to the Bonneville tailrace (Anderson 1998). The exact assumptions used to generate this estimate have not been reviewed by PATH at this point and no comparable spring/summer chinook survival estimates were presented.

In the absence of passage simulation modeling that would allow comparison of steelhead and spring/summer chinook survival, a very simple analysis to compare each species’ survival under recent conditions indicates that direct survival to below Bonneville Dam is at least as high for juvenile steelhead as it is for juvenile spring/summer chinook salmon under recent operations. The elements of this analysis are: 1) the range of inriver survivals estimated in Table 4.7.1-5; 2) the direct transport survival rate estimated above (98% for each species); and 3) the relative proportion of fish entering Lower Granite pool that are then transported.

Graves and Ross (1998) estimate that under recent operations a larger proportion of wild Snake River steelhead than wild Snake River spring/summer chinook salmon have been transported from the Snake River (Table 4.7.1-6). Estimates range from 3 to 23% more steelhead transported than chinook in recent years, depending upon fish guidance assumptions and annual operations. These estimates are based on the proportion of juveniles arriving at Lower Granite Dam and therefore over-estimate the proportion of Lower Granite Reservoir arrivals that are transported. Iwamoto et al. (1994) and Muir et al. (1995) suggest that most of the Lower Granite combined reservoir and dam reach mortality during the spring occurs at the dam, with little occurring in the reservoir. Some previous PATH analyses have considered 95% to be a conservative approximation of spring/summer chinook survival through Lower Granite Reservoir (Chapter 6 in Marmorek et al. 1996).

Table 4.7.1-6: Estimates of percentage of smolt arriving at Lower Granite Dam that have been transported from Snake River collector projects during the last three years of interim operations (Graves and Ross 1998). Results for two assumptions regarding fish guidance efficiency (FGE) are presented. In one, extended-length bar screens are assumed to have the same FGE as standard traveling screens (ESBS = STS). In the second, the FGE of extended-length screens is assumed to be higher (STS < ESBS). [H = hatchery, W = wild]

Year	**Snake River Steelhead**		**Snake River Yearling Chinook**		**Difference (SH-CH)**	
	STS = ESBS	STS < ESBS	STS = ESBS	STS < ESBS	STS = ESBS	STS < ESBS
1995	0.800 (H) 0.919 (W)	N/A	0.583 (H) 0.674 (W)	N/A	0.22H 0.23W	N/A
1996	0.550 (H) 0.641 (W)	0.550 (H) 0.641 (W)	0.341 (H) 0.422 (W)	0.460 (H) 0.597 (W)	0.21H 0.22W	0.09H 0.06W
1997	0.498 (H) 0.579 (W)	0.498 (H) 0.579 (W)	0.318 (H) 0.389 (W)	0.426 (H) 0.552 (W)	0.18H 0.19W	0.07H 0.03W

The above information can be combined as follows:

$$S_{DIRECT} = [T * S_{LGR} * S_{TRAN}] + [(1 - T * S_{LGR}) * S_{INRIVER}]$$

where S_{DIRECT} is direct survival to below Bonneville Dam; T is the proportion of fish arriving at Lower Granite Dam that are subsequently collected for transportation from all collector projects (4.1-8); S_{LGR} is the survival from the head of Lower Granite Reservoir to Lower Granite Dam (assumed to be 0.95 for this analysis -- see above); S_{TRAN} is direct survival of transported fish from collection until release (assumed to be 0.98 in this analysis - see above); and $S_{IN\text{-}RIVER}$ is direct survival of uncollected fish that migrate inriver (0.8 times estimates in Table 4.7.1-5 - see above). Combining the ranges of estimates for wild steelhead and spring/summer chinook in Tables 4.7.1-5 and 4.7.1-6 allows estimates of direct survival in 1995 and 1996 for wild smolts of each species, using comparable methods. Table 4.7.1-7 indicates that, under recent operations, direct survival to below Bonneville Dam during those years has been at least as high, and possibly somewhat higher, for juvenile steelhead as it has been for juvenile spring/summer chinook salmon.

Combined Transport and In-River Direct+Indirect Survival Under Current Operations

Analyses to this point assume that effects of the FCRPS end when smolts pass, or are released from transport, immediately below Bonneville Dam. Various indirect effects of the FCRPS have been proposed beyond this point, and several hypotheses have been articulated and evaluated by the PATH analytical group (Marmorek and Peters 1998a,b), but these have not been applied to steelhead. See Section 4.8 for discussion.

Table 4.7.1-7: Estimates of direct survival to below Bonneville Dam of transported and untransported wild Snake River steelhead and wild Snake River spring/summer chinook salmon. The purpose of this table is to compare relative survival of the two species in recent years, using similar techniques — it is not to make predictions regarding future survival. In-river survival of wild fish is from Table 4.7.1-5; transport proportions are from Table 4.7.1-6; and direct survival estimates are from Equation 1, with constants defined as in text. Complete information for wild fish in 1994 and 1997 is not available at this time.

Year	Steelhead			Spring/Summer Chinook			Difference in Direct Survival (SH-CH)
	In-River Survival	Transport Proportion	Direct Survival	In-River Survival	Transport Proportion	Direct Survival	
1994	0.8*0.171			0.8*0.234			N/A
1995	0.8*0.531	0.919	0.910	0.8*0.636	0.674	0.811	0.099
1996	0.8*0.872	0.641	0.870	0.8*0.908	0.422 to 0.597	0.828 to 0.870	-0.001 to +0.041
1997		0.579			0.389 to 0.552		N/A

Table 4.7.1-8: Summary of recent studies comparing survival of transported vs. inriver migrants for Snake River steelhead and Snake River spring/summer chinook salmon. All transported fish were barged, rather than trucked. 95 % Confidence Intervals in parentheses.

Study	Steelhead Transport:In-River Return Rate	Yearling Chinook Transport:In-River Return Rate	Difference (SH-CH)
Matthews et al. (1992) Mixed Hatchery/Wild	2.0 (1.4-2.7)	1.6 (1.01-2.47)	0.4
Harmon et al. (1995) Mixed Hatchery/Wild	2.1 (1.3-3.5)	2.4 (1.4-4.3)	-0.3

Transport studies that include information on survival through the estuarine and ocean environments provide one type of information that is useful in evaluating the indirect effects of transportation. Recent studies (i.e., since 1986) comparing transported and in-river migrating Snake River steelhead and Snake River spring/summer chinook survival indicate higher returns of transported than untransported juveniles for both species (summarized in Table 4.7.1-8). The transport:in-river return rate (T/I return rate) for Snake River steelhead and Snake River spring/summer chinook salmon in these studies varies slightly, but is generally similar for both species. Results of more recent transport studies, conducted in 1995 and 1996, will become available as adults return during the next three years.

Adult Fallback Rates and Fallback Mortality

Available information indicates high mortality for adult steelhead which fall back past dams through turbines. Mortality estimates for fallback of adult steelhead through turbines range from 22% to 57% (Buchanan and Moring 1986; Liscom et al. 1985; Wagner and Ingram 1973). PATH is not aware of information regarding the fallback mortality of adult chinook salmon through turbines. However, using a theoretical strike methodology to address the relationship between direct strike by a turbine blade and fish length, estimates of 41% and 49% mortality for 25- and 30-inch fish (respectively) are derived, similar to those observed for adult steelhead (Scott 1985).

A substantial percentage of adult salmon and steelhead passing dams have been observed to fall back through spillways at certain dams under certain conditions (Bjornn and Peery 1992). High fallback rates are usually associated with high river flows and spill, as well as the location of fishways exits relative to the spillways. In studies in which both adult chinook and steelhead were radio-tagged, fallback rates are generally similar for both species (Table 4.7.1-9). Fallback rates ranged from 0% to 38.9% and 0% to 50% (n = 4 for the 50% estimate) for chinook and steelhead, respectively. Liscom et al. (1979) concluded from several fallback studies that fallback rates can be high at times, but few fish are injured or die as a direct result of fallback; migration times are increased if the fish must reascend the dam.

Adult Passage Mortality

Cumulative passage mortality for adult steelhead migrating up the Columbia and Snake Rivers through eight mainstem dams can be substantial. One estimate of loss is calculated from the difference in adult counts between successive dams (after adjustment for legal harvest) and represents loss and mortality. Mortality can be caused by: effects of delayed migration (due to project or naturally-increasing summer temperatures), fallback through turbines, illegal harvest, delayed mortality from marine mammal predation, gas supersaturation, gillnet interactions, and disease.

Table 4.7.1-9: Estimates of adult fallback past Snake and Columbia River projects from radio-telemetry studies of both steelhead and chinook. * = Steelhead and chinook in this study were released 1,300 feet upstream of Lower Monumental Dam.

Project	Reference	Chinook Fallback Rate	Number Tagged	Steelhead Fallback Rate	Number Tagged	Difference (SH - CH)
Bonneville	Monan and Liscom (1975)	.389 (Summer)	18	.500	4	.111
	Liscom et al. (1978) in Bjornn and Perry (1992)	.022 (Spring)	90	.000	35	-.022
	Ross (1983)	.150 (Summer/Fall)	20	.050	20	-.100
	Ross (1983)	.000 (Fall)	12	.000	14	.000
Lower Monumental*	Liscom et al. (1985) in Bjornn and Perry (1992)	.094 (Summer/Fall)	32	.202	258	.108
Little Goose*	Liscom et al. (1985) in Bjornn and Perry (1992)	.077 (Summer/Fall)	13	.038	157	-.039

Apparent adult loss between dams may also be due to factors other than mortality of adults, such as:

- counting errors;
- double-counting fish that fall back and re-ascend ladders;
- straying; and
- tributary turnoff.

The combination of these effects has led to apparent adult passage losses between Bonneville Dam and Lower Granite Dam.

Another indication of adult passage loss (which excludes counting errors, double-counting fish that ascend ladders more than once, and straying or turnoff into tributaries) is data from radio tagging studies (Bjornn et al. 1992, 1994, 1995; Turner et al. 1984b; Liscom et al. 1978; Ross 1983; Monan and Liscom 1976). Based on these studies comparing passage of both chinook and steelhead, the combined passage loss of radio-tagged fish in the lower Snake and lower Columbia Rivers (which is applicable to Snake River steelhead) is estimated to be 20.8% (79.2% survival) (Table 4.7.1-10; Ross 1998). The 20.8% loss of radio-tagged Snake River steelhead appears to be a more representative estimate of mortality attributable to passage through the FCRPS than estimates based on dam counts. This estimate of steelhead survival (79.2%) is greater than the 74.8% survival of chinook salmon, estimated in an identical manner using information from the same studies.

Survival of adult steelhead may be related to water temperature, an effect that would influence steelhead to a greater degree than spring/summer chinook salmon. Anecdotal information suggests that during years with high Snake River temperature, a thermal block develops near the confluence of the Snake and Columbia Rivers, which may delay migration or result in increased straying (Matthews et al. 1992; Harmon et al. 1992).

Mortality of Downstream-Migrating Adults (Kelts)

Unlike chinook salmon, which die after spawning, steelhead may survive to spawn more than once. Adults that have spawned and are migrating back downstream to return to the ocean are referred to as “kelts.” Estimates of the number or proportion of steelhead that survive spawning are rare. In 1994, the only year for which kelt estimates at an FCRPS project have been published, 47 wild Snake River steelhead kelts passed downstream via the juvenile bypass system at Lower Granite Dam (Hurson et al. 1996). This corresponds to approximately 0.6% of the TAC (1997) estimates of the number of wild steelhead adults that passed Lower Granite Dam during either 1993 or 1994. It is possible that a higher proportion of spawners may have migrated downstream as kelts if they passed Lower Granite Dam through other routes (e.g., through turbines or spillway). The number of kelts that passed Lower Granite Dam through turbines in 1994 is unknown. Because of very limited spill, it is likely that few kelts passed via the spillway during 1994. The number of Snake River steelhead kelts passing other FCRPS dams is unknown, as are numbers of Upper Columbia River and Lower Columbia River steelhead passing FCRPS dams.

The mortality of kelts passing FCRPS projects has not been estimated. For those that pass through turbines, the mortality is likely to be at least as high as that estimated for upstream-migrating adults that fall back through turbines (see discussion above). It is unlikely that many kelts survive dam passage to spawn a second time.

Table 4.7.1-10: Estimates of adult survival past Snake and Columbia River projects from radio-telemetry studies, as summarized in Ross (1998).

Project	Reference	Chinook Project Survival	Average Chinook Project Survival	Steelhead Project Survival	Average Steelhead Project Survival	Difference (SH - CH)
Bonneville	Turner et al. (1984)	0.900 (Fall)		0.943 (Fall)		0.043
Bonneville	Liscom et al. (1978)	1.000 (Spring)		0.925 (Summer)		-0.075
Bonneville	Ross (1983)	0.872 (Spring)		0.952 (Summer)		0.080
Bonneville	Ross (1983)	0.943 (Fall)		1.000 (Fall)		0.057

Project	Reference	Chinook Project Survival	Average Chinook Project Survival	Steelhead Project Survival	Average Steelhead Project Survival	Difference (SH - CH)
Bonneville	Monan and Liscom (1976)	1.000 (Spring)		1.000 (Fall)		
Bonneville Mean (Lower Columbia River Steelhead Total FCRPS Reach)			0.943		0.964	0.021
The Dalles	Liscom et al. (1978)	0.985 (Spring)		1.000 (Summer)		0.015
The Dalles	Monan and Liscom (1976)	0.946 (Spring)		1.000 (Summer)		0.054
The Dalles Mean			0.966		1.000	0.034
John Day	Liscom et al. (1978)	1.000 (Spring)		1.000 (Summer)		0.000
McNary Estimate	Ross (1998)		0.970		0.988	0.018
Bonneville to McNary Mean (Upper Columbia River Steelhead Total FCRPS Reach Estimate)	Ross (1998)		Average Chinook Reach Survival 0.884		Average Steelhead Reach Survival 0.952	0.068
Four Snake River Projects	Bjornn et al. (1992)	0.870		0.813		-0.057
Four Snake River Projects	Bjornn et al. (1994)	0.810		0.870		0.060
Four Snake River Projects	Bjornn et al. (1995)	0.861		0.813		-0.048
Four Snake River Projects Mean			0.847		0.832	-0.015
Estimate for Eight-Dam System (Snake R. Steelhead Total FCRPS Reach Estimate)	Ross (1998)		Average Chinook Reach Survival 0.748		Average Steelhead Reach Survival 0.792	0.044

Prior to construction of most lower Columbia River and lower Snake River dams, the proportion of repeat-spawning summer steelhead in the Snake and Columbia Rivers was less than five percent (3.4% [Long and Griffin 1937]; 1.6% [Whitt 1954]). Repeat-spawning in winter steelhead (e.g., some populations of the Lower Columbia River steelhead ESU) was apparently as high as 12% (Long and Griffin 1937). Recently, in the lower Columbia River, summer steelhead populations that do not pass through any dams (Kalama Rivers – Lower Columbia River ESU) or that pass through only one dam (Klickitat River – Middle Columbia River ESU) have approximately seven percent and three percent proportions of repeat spawners, respectively (Howell et al. 1985, cited in Busby et al. 1996). Lower Columbia River ESU winter steelhead that do not pass through any dams (Cowlitz and Kalama Rivers) have approximately four percent and eight percent repeat spawners, respectively (Howell et al. 1985, cited in Busby et al. 1996).

4.7.2 Comparison of Steelhead vs Spring/Summer Chinook Survival Resulting from Proposed Management Actions

Management Action A1

The current operations described in Section 4.7.1 describe this action.

Management Actions A2 and A2'

Because steelhead have a higher fish guidance efficiency, more steelhead than spring/summer chinook salmon are affected currently transported under A1. One implication is that the proportional change in the percentage of steelhead transported under A2 and A2', relative to A1, may be less for steelhead than for chinook. This may make the beneficial effects of A2 and A2' (relative to A1) on **direct** survival less for steelhead than for spring/summer chinook. However, to the extent that detrimental effects of transportation on post-Bonneville mortality are assumed (e.g., "hydro" extra mortality hypothesis), the detrimental effects of additional transportation under A2 and A2' (relative to A1) would also be reduced for steelhead, compared to chinook.

Management Actions A3 and B1

Estimates of the inriver survival of Snake River steelhead through the FCRPS that may occur as a result of drawdowns or future project passage improvements are not presently available. However, it is likely that drawdowns will affect juvenile steelhead in a similar manner to Snake River spring/summer chinook salmon. For example, Table 4.7.2-1 indicates that survival of both yearling chinook and steelhead through free-flowing reaches of the Snake River above Lower Granite Dam are variable, with steelhead survival higher in three of the available years and yearling chinook survival higher in one year. Under the assumption that survival in a future drawn-down section of the Snake River will (after some equilibration period) be similar to survival in upstream free-flowing reaches, Table 4.7.2-1 suggests that steelhead direct survival through that reach may be between 7.5% lower to 12.5% higher than chinook survival, and will average approximately two percent higher. Effects of drawdowns on the indirect survival (i.e., below Bonneville Dam) of steelhead versus chinook cannot be described at this time. Presumably, effects of John Day drawdown also will be similar to effects of Snake River drawdown for the two species.

Table 4.7.2-1: Estimates of wild juvenile steelhead and wild yearling chinook survival in free-flowing river sections above Lower Granite Dam. Estimates of survival from Whitebird (Salmon Trap) to Lower Granite (LGR) tailrace and from the head of LGR Reservoir (Snake Trap) to LGR dam from Table 24 and Appendix Tables E7-E9 of Smith et al. (1998). These estimates do not necessarily reflect survival of the same population of fish through the two reaches, so the method of removing effects of LGR Reservoir and dam passage in Column (3) is imperfect. Bold estimates are those which appear to be extremely high, possibly due to the methodology used in this table and, in the case of the 1996 estimate, due to high standard error associated with the original reach survival estimate.

	(1) Whitebird (Salmon River) to LGR Tailrace (233 km)		(2) Head of LGR Reservoir to LGR Tailrace (52 km)		(3) = (1)/(2) Estimated Survival Whitebird to Head of LGR Reservoir (181 km)		(4) = Mean per-km survival from (3) raised to 210th power Estimated Survival Through Snake Drawdown Section (210 km)		Difference (SH - CH)
Year	**Steelhead**	**Chinook**	**Steelhead**	**Chinook**	**Steelhead**	**Chinook**	**Steelhead**	**Chinook**	
1993	0.832	0.832	0.898	0.839	0.927	0.992	0.915	0.990	-0.075
1994	0.75	0.788	0.836	0.894	0.897	0.881	0.882	0.864	0.018
1995	0.892	0.863	0.955	0.944	0.934	0.914	0.924	0.901	0.023
1996	0.967	0.882	0.945	0.964	1.023	0.915	1.027	0.902	0.125

Management Actions A6 and A6'

While no attempt was made to evaluate A6, the inriver option, for steelhead, an evaluation was conducted for the spring/summer chinook stock. As noted elsewhere, steelhead and spring/summer chinook behave and survive similarly during passage through the hydropower system and would therefore be expected to have similar relative responses to the proposed management actions, including A6/A6'.

The A6 option was intended to depict mainstem operations in which survival was enhanced by several means. As modeled, approximately one Million Acre Feet (MAF) was added to the current 427 Thousand Acre Feet (KAF), which served to increase reservoir survival. Dam survival was improved, first, through additional spill made possible by the addition of gas abatement structures intended to reduce the incidence and severity of Gas Bubble Trauma. Dam survival was also increased by simulating the potential improvements made possible by the installation of surface bypass structures at all projects.

Results of an assessment of A6 conducted for spring/summer chinook indicated that even with fairly optimistic assumptions about surface bypass systems, well in excess of results of prototype experiments to date, in-river improvements were generally inadequate to increase survival beyond that achieved through the current transportation based system. It is unrealistic to assume that the in-river improvements simulated in A6 would approach those projected for A3, the drawdown of four Snake River dams to natural river.

4.8 Detailed Evaluation of Point #6: Action Will Have Similar Survival Effect on Steelhead and Chinook Outside of the Hydrosystem

One implicit assumption of this comparative analytical approach is that an equivalent incremental change in direct hydrosystem survival for steelhead and spring chinook, as a result of a proposed management action, will result in an equivalent incremental change in SAR. Factors other than direct hydrosystem survival that are components of Escapement SAR include: harvest, indirect effects of juvenile passage above Bonneville (mortality caused by some upstream factor or factors, such as hydrosystem injury and/or hatchery-related disease transmission, that is expressed below Bonneville Dam), and other estuarine and ocean mortality. This last factor is presumed to consist primarily of natural mortality, but may also include some human-caused effects, such as estuarine habitat modification (e.g., man-made islands supporting large predacious bird colonies).

The only component of SAR that can be empirically estimated, besides direct hydrosystem survival, is harvest. The more detailed spring/summer chinook analysis of patterns in stock and recruitment has indicated that there is a source of "extra mortality," which cannot be explained by estimated direct mortality and the underlying stock-recruitment relationship (e.g., $1-e^{[-(m-M)]}$ in Marmorek and Peters 1998a,b). Various hypotheses have been proposed to account for the causes of "extra mortality" and the way in which this mortality would be expected to change in the future as a result of proposed management actions. It may be related to indirect effects of upstream passage, which are expressed as mortality below Bonneville Dam, or it may be related to differential ocean survival of various stocks, and changes in the pattern of that survival over time. These hypotheses have considerable influence on the results, so it is important to examine the assumption that these "extra mortality" hypotheses can be applied equally to Snake River steelhead and spring/summer chinook salmon.

This section examines the three main components of SAR in turn, attempting to evaluate the likelihood that a proposed management action will have a similar effect on each component for steelhead and spring/summer chinook.

4.8.1 Harvest

Ocean harvest of both steelhead and spring/summer chinook salmon is effectively non-existent. Recent in-river harvest rates for wild Snake River steelhead are approximately three times higher than harvest rates for wild Snake River spring/summer chinook (Table 4.8.1-1, based on TAC 1997, 1998; Appendix B, Table B.3-4).

Table 4.8.1-1: Comparison of wild Snake River steelhead and spring/summer chinook harvest rates in recent years (TAC 1997,1998). "Combined" harvest includes commercial harvest in Zones 1-6, sport harvest, and tributary harvest

Return Year	Run Size (to Columbia River mouth)	Combined Harvest	Harvest Rate
Spring/Summer Chinook			
1991	8815	626	0.071
1992	17957	1043	0.058
1993	11602	730	0.063
1994	2530	269	0.106
1995	2363	127	0.054

Return Year	Run Size (to Columbia River mouth)	Combined Harvest	Harvest Rate
Average			0.070
Steelhead			
1991	16304	2898	0.178
1992	19114	3481	0.182
1993	39006	9548	0.246
1994	32219	7758	0.241
1995	21069	5014	0.238
Average			0.216

4.8.2 Other Components of SAR

Two general lines of evidence suggest that the other components of smolt-to-adult survival may respond similarly to management actions for steelhead and spring/summer chinook. The first is similarity in trends among the two species. As discussed in Section 4.7, direct hydrosystem survival of the two species appears to be very similar, and trends in direct reach survival between the two species are nearly identical (Table 4.7.1-5, Figure 4.7.1-2). Additionally, trends in SAR, adjusted for harvest, are also nearly identical over most (but not all – see 1977-1988) of the time period evaluated for the two species (Figure 4.7.1-1, from data in Petrosky 1998a; Petrosky and Schaller 1998; Williams 1998; Williams et al. 1998). This suggests that the components of SAR not accounted for by direct hydrosystem survival and harvest must also trend nearly identically for each species. Therefore, to the extent that hydrosystem management actions affect (or do not affect) the remaining survival components for Snake River spring/summer chinook salmon, they probably also do so for Snake River steelhead.

"Natural" Estuarine and Ocean Mortality

The rate of natural mortality of Snake River steelhead and spring/summer chinook salmon through the Columbia River estuary and ocean is unknown. Some factors suggest that survival may be similar for each species through this life stage. For example, juveniles reach the estuary at a similar time and therefore may experience similar predator and prey fields as well as physical oceanographic conditions. Both species migrate through the ocean in deeper water than other salmonid species, making them less vulnerable to harvest by coastal fleets and suggesting that ocean mortality rates are more similar for these species in comparison to each other than in comparison to other salmon species. Both species spend 1-3 years in the ocean, with most steelhead maturing after one year and most chinook after two years. If the rate of natural mortality is similar for the two species, it will accumulate over a longer period of time for spring/summer chinook salmon, possibly resulting in higher survival of steelhead in the ocean

However, the life history of steelhead, compared to spring/summer chinook salmon, suggests that some caveats must be applied to the assumption that the two species experience similar mortality through this life stage. Juvenile steelhead tend to spend a longer time rearing in tributaries prior to seaward migration, compared to spring/summer chinook, so at the time of juvenile migration steelhead have experienced higher freshwater mortality but the survivors are approximately four times larger than spring/summer chinook salmon (Williams et al. 1998 – mean weight of wild steelhead at Lower Granite Dam in 1998 was 58.7 g, compared to 12.5 g for spring/summer chinook) and therefore may experience lower mortality through other life stages, such as estuarine and ocean migration.

A climatic "regime shift" hypothesis proposes that Snake River spring/summer chinook estuary and ocean survival has decreased substantially since the late 1970's as a result of changing oceanographic conditions, resulting in an increase in the "extra mortality" term during recent years. Because downstream spring chinook stocks have not experienced an equivalent increase in "extra mortality", this hypothesis proposes that Snake River chinook experience different ocean mortality than other Columbia River basin spring chinook stocks. That is, for this hypothesis to be true, the unique migratory characteristics of Snake River spring/summer chinook salmon would have to be more like those of Snake River steelhead than like those of lower Columbia River spring/summer chinook salmon. The "extra mortality" also affects fish that were transported or migrated in-river as juveniles differentially. The hypothesis proposes that the "extra mortality" will again decrease following the next oceanographic regime shift. Evidence for or against this hypothesis is very limited, largely due to the paucity of ocean coded-wire tag recoveries for Snake River spring/summer chinook salmon. This same situation is true for Snake River steelhead. Therefore, to the extent that this hypothesis applies to spring/summer chinook, it may also apply to steelhead.

Indirect Effects of Juvenile Passage Above Bonneville Dam

Experiences of juvenile salmonids during passage above Bonneville Dam may be expressed as mortality below Bonneville Dam. Direct empirical estimates of this effect are not available, but at least three hypotheses previously described in PATH suggest that the Snake River spring/summer chinook salmon "extra mortality" term may be caused by above-Bonneville experiences.

A "BKD" hypothesis suggests that some factor, such as bacterial kidney disease, is experienced by fish above Bonneville Dam and this experience results in mortality below the dam. This situation developed in the mid- to late-1970s, resulting in higher "extra mortality" of Snake river spring/summer chinook since that time. Higher below-Bonneville "BKD-caused" "extra mortality" is experienced by transported than non-transported fish. The hypothesis proposes that the increase "extra mortality" experienced in recent years is not likely to decrease in the future, regardless of any change in management actions evaluated by PATH in the spring/summer chinook analysis (Marmorek and Peters 1998a,b). The extent to which this hypothesis explains "extra mortality" in steelhead is difficult to evaluate because an explicit mechanism has not been identified. If that mechanism is, in fact, BKD, then the effect of this hypothesis may be less on steelhead than on spring/summer chinook, because the infection and mortality rates are lower in steelhead.

A "Hatchery" hypothesis suggests that interactions between wild and hatchery fish, particularly hatchery steelhead, are detrimental to wild spring/summer chinook salmon because of disease transmission, stress/predation vulnerability, and reduction of available food. These interactions occur primarily above Bonneville Dam but mortality may not be expressed until fish pass Bonneville Dam. This "extra mortality" is higher for transported than non-transported fish, possibly due to crowding in collection facilities. Large increases in hatchery releases, beginning in the late 1970s, caused the increased "extra mortality" observed in recent years and this "extra mortality" will not decrease unless there is a reduction in hatchery releases. Some components of this hypothesis probably apply equally well to wild steelhead and wild spring/summer chinook, such as disease transmission and reduction of food availability. Other components, such as increased vulnerability to predation caused by stress, are probably less likely for wild steelhead, because agonistic interactions among hatchery and wild steelhead probably cause less stress than interactions among hatchery steelhead and wild spring/summer chinook.

A "Hydro" hypothesis suggests that stress and injury caused by passage through dams, barges, and reservoirs results in mortality that is expressed below Bonneville Dam. This "extra mortality" increased in the mid-1970s, after completion of new dams on the Snake River. The "extra mortality" is higher for transported than non-transported fish. To the extent that this hypothesis explains "extra mortality" in spring/summer chinook salmon, it probably also does so for steelhead because both species experience similar effects of hydrosystem passage.

4.9 Conclusions

4.9.1 Conclusions From Existing Information

In Section 4.1 (Point #7) a number of assumptions, upon which the analysis in this report relies, were presented. Most of the information presented in this report has been organized to evaluate the validity of those assumptions. The following summarizes our conclusions regarding each one.

Do spring/summer chinook management actions that result in an acceptable probability of being above survival threshold levels and reaching recovery levels correspond to historical smolt-to-adult survival rates (SAR)?

> We conclude that the answer is "yes" for the 100-year survival threshold goal and the 48-year recovery goal. However, Escapement SARs somewhat higher than those observed historically may be required to ensure that populations remain above survival thresholds over the next 24 years (Figure 4.3-1).

Is the incremental change between current and historical SAR for steelhead less than or equal to the incremental change for spring/summer chinook?

> We conclude "yes", based on the information presented in Tables 1 and 2. We note that choice of "historical period" for Snake River steelhead is subject to judgement and choice of alternative years could influence the necessary incremental change. However, even with certain alternative time periods for which historical estimates exist, which were discussed by the steelhead work group, this conclusion would not change. Similarly, the conclusion is not affected by choice of an SAR metric.

Is the management action likely to have a similar effect on both Snake River steelhead and spring/summer chinook hydrosystem survival?

> We conclude "yes". Mainstem survival characteristics are similar between the two species and these similarities extend to reach survival estimates, dam passage estimates, fish guidance efficiencies (FGEs), and Transport/In-River survival ratios (TIs).

Is the management action likely to have a similar effect on both Snake River steelhead and spring/summer chinook survival outside of the hydrosystem?

> We tentatively conclude "yes", but note that this assumption is very difficult to evaluate using our qualitative approach. The main evidence supporting this conclusion is the similarity in temporal patterns of SAR and in-river passage survival. This suggests that trends in non-passage components of SAR are also fairly synchronous, responding similarly to past management actions and environmental conditions. We note that the distribution of mortality throughout each species' life cycle is not expected to be identical, so responses to management actions also may not be identical. Of particular note are the higher tributary mortality rates likely for steelhead because of their extended residence time and the significantly higher harvest rates experienced by steelhead, compared to spring/summer chinook.

Based on conclusions for these assumptions, we further conclude that, if a management action is likely to meet survival and recovery standards for Snake River spring/summer chinook salmon, it is also likely to meet survival and recovery standards for Snake River steelhead.

However, if an action fails to result in an acceptable likelihood of survival and recovery for spring/summer chinook, this method does not address whether steelhead have an acceptable or unacceptable likelihood. Because it appears that the incremental survival change needed for steelhead is less than that needed for chinook (Section 4.6), it is possible that some actions that are unacceptable for chinook may be acceptable for steelhead.

4.9.2 Future Tasks for FY99

Development of a more quantitative approach to analysis of the effects of proposed actions on steelhead would require considerable additional work. Some of the tasks that would be useful include the following.

High Priority

- Development of passage model inputs, based upon further review of empirical data by PATH Hydro work group – e.g., FGE, historic reach survival estimates, etc.
- Passage model development and retrospective model runs 1964-1994. These would include estimates of direct passage survival, system survival, proportion below Bonneville transported, etc.
- Passage model runs—prospective (A1, A2, A3, B1, A6/A6’, A2’ + others from IT), estimates of system survival prospective to retrospective ratios
- Prospective life cycle modeling. The exact method would have to be determined. Possibilities include: SAR based; as additional “stock” with unique passage model survivals in spring/summer BSM; or qualitative comparisons to spring/summer responses.
- Examine SAR sensitivity analyses suggested by Paulsen and Giorgi (1998).
- Conduct detailed review of the pros and cons of using the two alternative SAR definitions (Escapement SAR and Escapement+Harvest SAR) for various purposes, including definition of biological requirements for ESA jeopardy standards.
- Evaluate feasibility of estimating SARs for A-run and B-run Snake River steelhead.

Moderate Priority

- Retrospective habitat vs. steelhead [and chinook] parr density evaluation (Bill Thompson—in progress, nearing completion)
- Chapter 9 (Marmorek and Peters 1996) spawner-to-smolt evaluation for steelhead - feasibility not certain (multiple smolt ages)
- Evaluation of trends in parr density (1985-1997) for different steelhead population groups, analogous to ln(spawner:spawner) ratios. Address replacement at population level, and by geographic or A-run/B-run stock groups.

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5.0 Sockeye

Less research has been directed toward Snake River sockeye than toward other Snake River salmonids. Because the life-cycle of sockeye salmon, with its variable freshwater residency, is so complex, and because there is relatively little field data available, PATH has been unable to conduct a data-intensive assessment of historical abundance patterns. However, PATH has begun a **preliminary** assessment of the Snake River stock's current status, preparatory to evaluating the effects of potential management actions.

5.1 Historical Population Trends

Historically, Snake River sockeye were produced in the Stanley Basin of Idaho's Salmon River in Alturas, Pettit, Redfish, Yellowbelly, and Stanley Lakes and may have been present in one or two other Stanley Basin lakes (Bjornn et al. 1968). Elsewhere in the Snake River Basin, sockeye salmon were produced in Big Payette Lake on the North Fork Payette River and in Wallowa Lake on the Wallowa River (Evermann 1895, Toner 1960, Bjornn et al. 1968, and Fulton 1970).

Bjornn et al. (1968) concluded that there were no reliable estimates of the number of sockeye spawning in Redfish Lake around the turn of the century. These authors stated their belief that, in the 1950s and 1960s, Redfish Lake was the only lake in Idaho still used annually by sockeye salmon for spawning and rearing. Hassemer et al. (1997) estimated that historic habitat availability for sockeye within Idaho (i.e., excluding Wallowa Lake, Oregon, from the Snake Basin total) was 3,719 hectares, compared to current habitat of 944 hectares.

At the time of listing under the Endangered Species Act (November 20, 1991; FR 56 No. 224), sockeye were produced only in Redfish Lake. In most years, agricultural diversions using all the water in Alturas Lake Creek prevented adult sockeye from migrating upstream. Thus, production in Alturas Lake had been precluded since early in the century. Alturas Lake was chemically treated to control rough fish in 1961 and 1962. Migration barriers, constructed to prevent the immigration of warmwater fishes precluded the production of sockeye in Pettit, Stanley, and Yellowbelly Lakes.

Beginning in 1910, access to Stanley Basin Lakes was seriously impeded by construction of Sunbeam Dam, 20 miles from Redfish Lake Creek on the mainstem Salmon River. The original adult fishway, constructed of wood, was ineffective in passing fish over the dam (Kendall 1912; Gowen 1914). That fishway was replaced with a concrete structure in 1920 that successfully passed sockeye salmon during at least one year. Some argue that the dam represented a complete barrier to upstream passage for enough years that the original anadromous run was eliminated (Chapman et al. 1990). However, eyewitness accounts (Jones 1991) document adult sockeye spawning in Redfish Lake during a number of years prior to and immediately after partial removal of the dam in 1934, and Parkhurst (1950) reported sockeye spawning in the lake during 1942.

Escapement of sockeye salmon to the Snake River has declined dramatically in recent years. Counts made at Lower Granite Dam have ranged from 531 in 1976 to zero in 1990. The Idaho Department of Fish and Game (IDFG) enumerated adult sockeye at a weir in Redfish Lake Creek during 1954 through 1966 (Bjornn et al. 1968). The number of adults counted by IDFG varied from 4,361 in 1955 to 11 in 1961.

The IDFG operated a temporary weir in Redfish Lake Creek during 1985 through 1987 (Kiefer et al. 1991). Total escapement in these years was 12 in 1985, 29 in 1986, and 16 in 1987 (Figure 5.1-1). Spawning ground surveys in Redfish Lake during 1988 identified four adults and two redds. One adult sockeye, one redd, and a second potential redd were observed in the lake during 1989. No redds or adults were observed in 1990. Since 1991, all adult sockeye returning to Redfish lake have been trapped at the weir and taken into a captive broodstock program (Pravecek and Johnson 1997; Kline and Lamansky 1997). Returns for 1991 through 1998 were 4, 1, 8, 1, 0, 1, 0, and 1 adults, respectively.

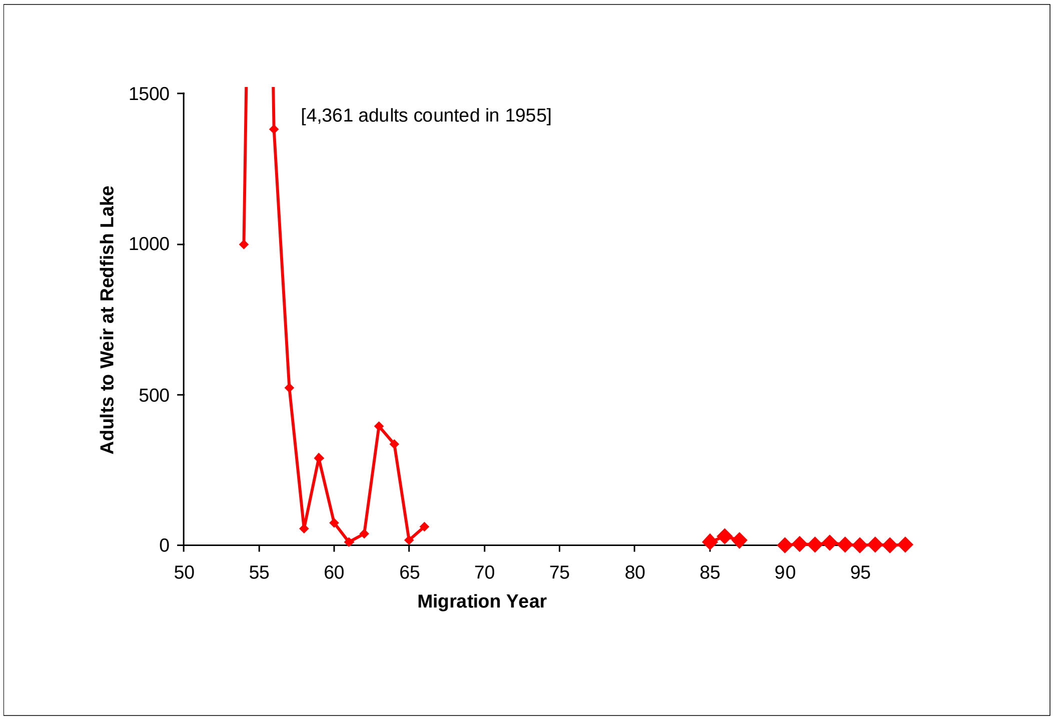

Figure 5.1-1: Sockeye salmon counts at Redfish Lake, Idaho (Kiefer et al. 1991).

Although variable, historical mainstem harvest rates on sockeye were high before the completion of the hydrosystem. Mainstem harvest rates averaged 0.40 (range 0 to 0.86) before and 0.06 (range 0 to 0.31) after 1974 (WDFW and ODFW 1998) (Figure 5.1-2).

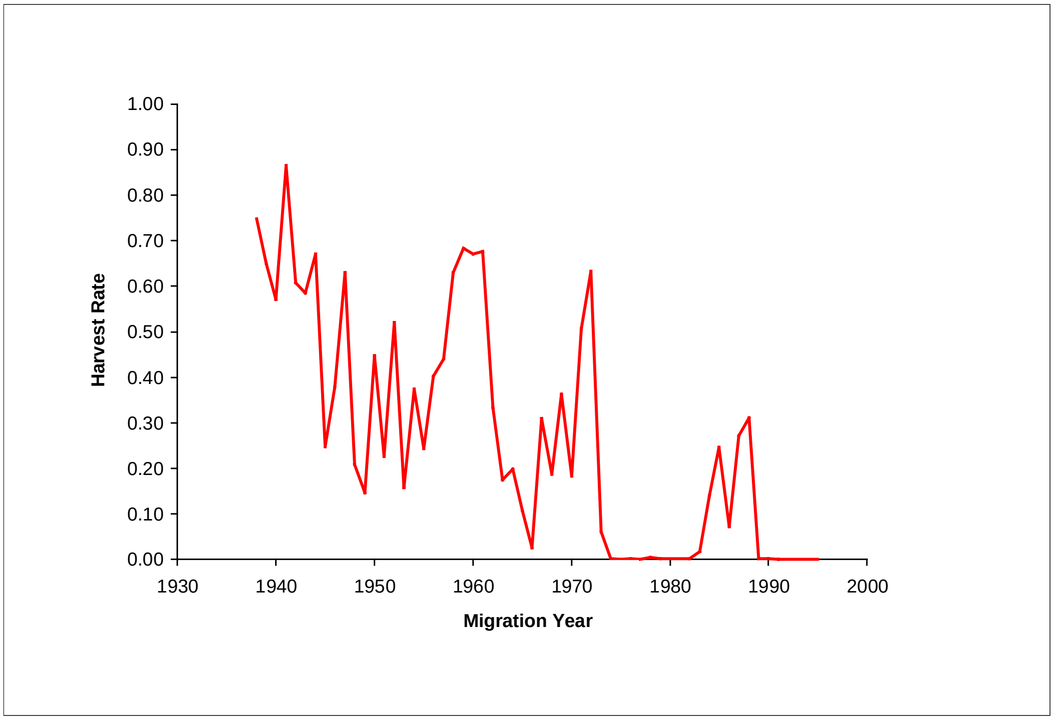

Figure 5.1-1: Mainstem harvest rates for sockeye salmon, Zones 1-6 (WDFW and ODFW 1998).

5.2 Factors for Decline

In the final rule listing Snake River sockeye salmon as an endangered species, NMFS listed the following factors affecting the species' abundance:

- Hydropower development;
- Water withdrawal and storage, irrigation diversions, and blockage of habitat for agricultural purposes;
- Overutilization by commercial fisheries in the lower Columbia River and harvest on the spawning grounds; and
- Predation in impounded reservoirs and in the estuary and nearshore ocean.

Exposure to bacterial, protozoan, viral, and parasitic organisms may also have played a role, but the effect of disease on Snake River sockeye has not been documented.

5.2.1 Correspondence of Decline with Hydropower Development

To date, PATH has explored some preliminary hypotheses regarding the effect of FCRPS hydropower development on historical abundances of Snake River sockeye. Sockeye abundance declined steeply following the period from 1968 through 1975 during which the number of Snake River hydropower dams increased from four to eight (see Ice Harbor Dam counts, Figure 5.2.1-1). A partial time series on adult returns to Redfish Lake indicates that the mean return from 1954 through 1964, a period when two to four dams were in place, was 769 sockeye. The mean return from 1985 through 1997 (with three years missing), when eight dams were in place, was seven sockeye. Smolt-to-Adult-Return (SAR) data are also available for the two periods, the period before intensive dam construction (outmigration years 1955 through 1964) and a recent period (outmigration years 1991 through 1996). The mean SAR for the preconstruction period, 0.8%, declined to 0.07% in the latter period. The SARs represent the survival rates of smolts from Redfish Lake to adult returns to the spawning grounds.

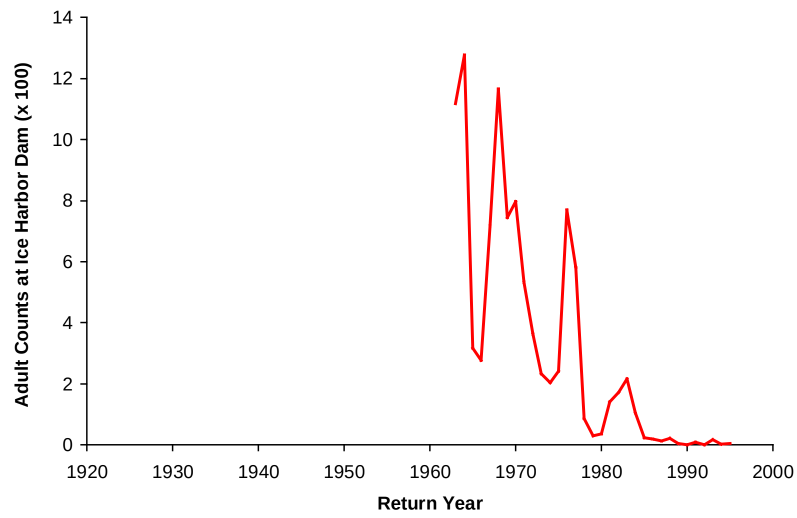

Figure 5.2.1-1: Adult sockeye salmon counts at Ice Harbor Dam (WDFW and ODFW 1998).

As with other Snake River species evaluated by PATH, this correspondence of dam development with the sockeye decline is confounded by other trends during the same period. Ocean environmental conditions changed, beginning in the late 1970s, as did the quantity of hatchery salmonid production. Mechanisms associated with both of these coincidental trends have been hypothesized as alternative, or at least contributory explanatory variables associated with declines of other Snake River salmonids.

In-River Juvenile Survival

In-river survival of juvenile sockeye salmon is assumed to be similar to that of Snake river spring/summer chinook, due to their similar size and timing. However, one factor suggests that survival of in-river migrant sockeye may be lower than that of Snake River spring/summer chinook salmon. Descaling rates for sockeye at lower Snake River and McNary dams may provide one of the mechanisms for mortality resulting from dam passage. Descaling rates for the period 1981 through 1997 are shown in Table 5.2.1-1.

Table 5.2.1-1: Descaling rates for Sockeye / Kokanee for lower Snake River and McNary dams.

Date	Stock Origin	Lower Granite	Little Goose	Lower Monumental	McNary	Notes
1997	Hatchery	9.9	0	13.9	9.7	
	Wild	24.5	10.7	14.1	18.7	
1996	Hatchery	3.8	5.3	6.7	11.6	
	Wild	18.4	14.8	5.9	11.5	
1995	Hatchery	3.2	9.4	4.8	5.7	
	Wild	30.1	15.7	13.6	18.3	
1994	Hatchery				7.8	
	Wild	12.5	15.1	21	12.4	
1993	Hatchery			26.6	2.9	
	Wild	27.3	11.1		8.5	
1992	Combined	2.3	6.6		13.1	
1991	Combined	0.5	5.9		10.8	1
1990	Combined		10			
1989	Combined				16.8	2
1988	Combined				10.4	
1987	Combined				10.9	
1986	Combined				21.1	
1985	Combined				8.8/3.0	3
1984	Combined				10.8	
1983	Combined				9.8	
1982	Combined				14.6	
1981	Combined				5.7-31.4	4

Notes:

1. 1993 – 1997 reported in annual reports of the Juvenile Fish Transportation Program. Numerous authors. U.S. Army Corps of Engineers, 1995 – 1998.
2. Pre-1993 summaries reported in annual reports of the Fish Transportation Oversight Team, FY81 through FY92. NOAA Technical Memoranda, NMFS F/NWR-2, -5, -7, -11, -14, -18, -22, -25, 27, -29, -31, and –32, respectively, 1981 through 1992.
3. Descaling criteria, developed by the Fish Transportation Oversight Team, changed in 1985. Criterion = 3.0 during earlier period; raised to 8.8 after 1985.
4. Range of descaling rates is based on eight days of sampling during May (pers. comm. C. Pinney [Corps of Engineers, Walla Walla District] to E. Weber, Fishery Biologist [Columbia River Intertribal Fish Commission]).

These data, when compared with similar estimates for steelhead and spring/summer chinook in Table 4.1-5, indicate that descaling rates are substantially higher for sockeye than for other salmonids for which data are available (Table A.2.3-3 in Marmorek and Peters 1998). Sockeye descaling rates did not decline when extended-length screens were installed at Lower Granite (1995 and 1996) or Little Goose (1997) dams. For years and projects where comparisons are possible, wild sockeye appear to experience greater descaling rates than those observed for hatchery sockeye. Although no estimates of juvenile sockeye salmon reach survival are available, it is reasonable to assume that sockeye salmon reach survival is lower than that of other Snake River yearling salmonids.

Partial support for the hypothesis that descaling causes relatively high in-river mortality of Snake River sockeye salmon can be obtained through comparisons with the mid-Columbia system. Although the evidence is restricted to correlation, declines in Snake River stocks, which pass through eight dams with screened passage facilities, have been more severe than those of Columbia River stocks, which pass through seven to nine dams, only four of which have screened bypass systems.

In the mid-Columbia River, mean escapement over Priest Rapids Dam was 121,000 sockeye in the five years (1952 through 1956) prior to intensive dam construction (WDFW and ODFW 1998) (Figure 5.2.1-2). Escapement fell to 55,800 in the period after construction (1969 through 1997), a 54% decline. In the Snake River, mean escapement over Ice Harbor Dam for the five years prior to intensive dam construction (1963 through 1967) was 741 sockeye; escapement fell to 99 sockeye after construction (1976 through 1997), an 86% decline (WDFW and ODFW1998). That is, sockeye migrating through four additional dams, without screened bypass systems or transportation, in the mid-Columbia reach were returning at a higher rate than sockeye migrating through three additional dams in the Snake River with screened bypass systems and transportation.

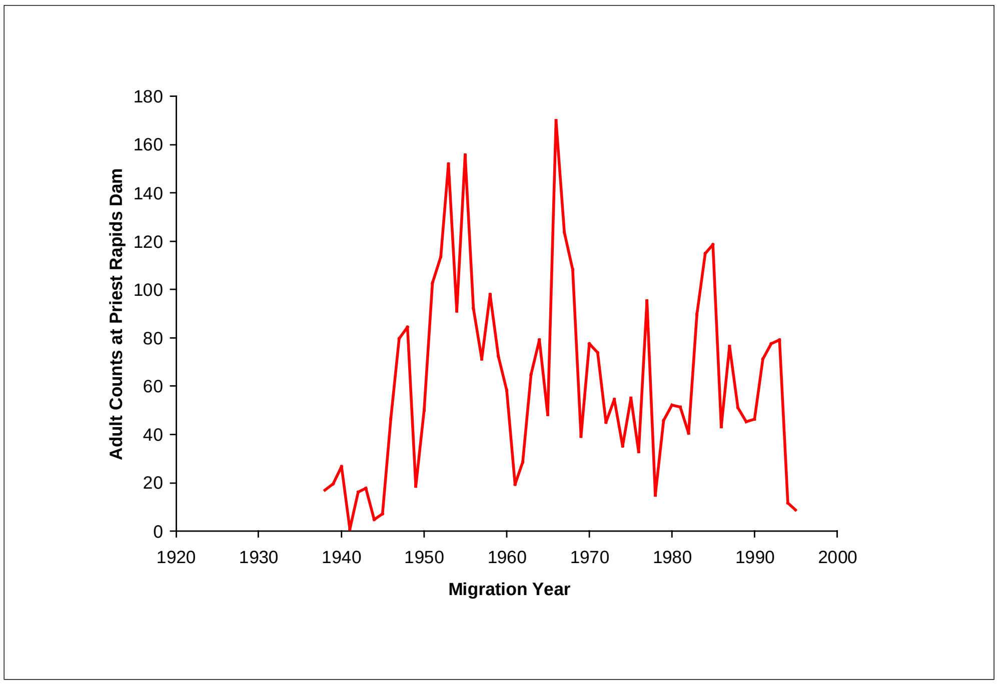

Figure 5.2.1-2: Adult sockeye salmon counts at Priest Rapids Dam (WDFW and ODFW 1998).

Transport Survival

Although most Snake River sockeye are presumably transported under current operations, no studies have been conducted to evaluate either direct or indirect transport survival. No information is available regarding relative SARs of transported and non-transported Snake River sockeye salmon.

Adult Passage

PATH has not yet reviewed passage survival data for adult sockeye salmon compared to data for other Snake River salmonids.

5.3 Potential Responses to Proposed Management Options

As noted, PATH has only begun assessing historical reasons for the decline of Snake River sockeye. However, if high rates of descaling, which appear to be associated with bypass screens, are a primary source of injury and mortality, options that rely on screens, such as A1 and A2, would be less likely to lead to recovery for sockeye salmon than for spring/summer chinook salmon. PATH has not yet addressed the likelihood of sockeye recovery under A3 or B1 relative to other species or evaluated the relative performance of the different options. Also, PATH has not yet evaluated the effectiveness of the captive broodstock program and its potential effect on survival and recovery of Snake River salmon. Further discussion of the effects of proposed management alternatives will be provided in PATH reports published during FY99.

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6.0 Experimental Management

6.1 Background

PATH grew out of previous efforts by various power regulatory agencies and state, federal, and tribal fisheries agencies to compare and improve the models used to evaluate management options intended to enhance recovery of ESA listed Snake River salmon. By 1994, an independent Scientific Review Panel concluded that rather than further analyzing model behavior, it would be more fruitful to test key hypotheses, particularly those related to the distribution of survival over the life span; the effects of flow on survival; and the benefit of transportation. This conclusion was formalized in the NMFS 1995 Biological Opinion on the Federal Columbia River Power System (page 124, recommendation 17).

PATH was therefore structured as a rigorous program of formulating and testing hypotheses. It is intended to identify, address and (to the maximum extent possible) resolve uncertainties in the fundamental biological issues surrounding recovery of endangered spring/summer chinook, fall chinook, and steelhead stocks in the Columbia River Basin. PATH's objectives are to:

1. determine the overall level of support for key alternative hypotheses and propose other hypotheses and/or model improvements that are more consistent with these data (retrospective analyses);
2. assess the ability to distinguish among competing hypotheses from future information, and advise institutions on research, monitoring and adaptive management experiments that would maximize learning; and
3. advise regulatory agencies on management actions to restore endangered salmon stocks to self-sustaining levels of abundance (prospective analyses).

PATH has done a lot of work on objectives 1 and 3, but much less work on objective 2. In past reviews of PATH products, the members of the Scientific Review Panel (SRP) have commented repeatedly on the need for an experimental-management approach to resolving key uncertainties (FY96, FY97, and FY98 SRP reviews). At the Weight of Evidence workshop held September 8-10, 1998, the SRP members strongly advised against delaying the 1999 decision due to uncertainty and again recommended using an experimental management approach (SRP 1998). They described two strategic experimental-management alternatives and discussed potential tools for the evaluation of experimental-management designs. Though the SRP provided their best judgements to weight alternative hypotheses on key uncertainties, they stressed that these weightings were not intended to replace research, monitoring, or experimental management actions that could, if carried out properly, produce data that would in time narrow these uncertainties further.

In this chapter we briefly introduce experimental management as it relates to PATH, provide some examples of how this approach could be helpful to the region, and describe what work remains to be done. A more detailed plan for work in FY99 is under preparation.

6.2 Experimental Management and PATH

The retrospective analyses and Weight of Evidence process completed by PATH have made considerable progress in reducing both the number of remaining uncertainties and their magnitude. Despite this progress, considerable uncertainty remains. This is not unusual. Forecasts of future results of management actions are often uncertain due to features common to all ecosystems: complex ecological responses to natural and human disturbances, variability over space and time, and inadequate knowledge of these systems. These uncertainties create biological risks, which have economic and social consequences. Ignoring these uncertainties (i.e., using only the best estimates of conditions), or using the existence of uncertainties as a rationale for maintaining the status quo, is not likely to improve the situation. The decision analysis approach undertaken in PATH explicitly and quantitatively considers the implications of uncertainties, so as to provide the best possible support to informed decisions.

A further improvement on the decision analysis approach taken to date is to choose those actions that attempt to both maximize the ability to achieve conservation and recovery objectives, and concurrently **learn something to improve future management**. This is called experimental or adaptive management (Walters 1986). It recognizes that uncertainties are unavoidable and that action cannot wait for uncertainties to be eliminated. The best experimental management approach to resolving uncertainties is called "active adaptive management" (Walters 1986). In an active adaptive management framework, resource managers implement deliberate experimental changes to a system to provide the contrast in treatments necessary to test or refine key management hypotheses. These changes are implemented in a spatial and/or temporal pattern that will reduce the confounding of management effects with other simultaneous events such as climate change. This approach increases the speed and chance of detecting effects if they exist and increases the precision and accuracy of estimated effects. This leads to stronger inferences about the effects of management actions, and increases the rate at which managers learn about the system and can improve or change management practices to best meet management objectives.

In contrast to active adaptive management, passive adaptive management is where managers wait for unplanned phenomena to generate system perturbations. A passive approach may increase the time required to obtain results. While the phrase "adaptive management" has often been used in Columbia Basin planning documents (e.g., the 1994 NPPC Fish and Wildlife Plan), actions have generally been taken to improve stocks without a strong means of evaluating their effectiveness, or formally evaluating the key uncertainties that determine that effectiveness (McConnaha and Paquet 1996). Under this approach, confounding is likely to be severe and researchers would be unable to identify which of many possible factors was responsible for observed effects. In general, past actions on the Columbia River are examples of passive adaptive management. This has contributed to the difficulties of disentangling potential causes of past population declines. For example, both the percent of fish transported, and the number of hatchery fish added to the system increased since the mid 1970's (Figure 6.2-1). This makes it very difficult to determine the relative importance of these factors in causing historical changes to salmon populations. A deliberately planned sequence of years with contrasting conditions in transport / hatchery factors would have been much more informative, and made PATH's current job easier. In addition to temporal contrasts, a planned design of spatial contrast of stocks (e.g. contrasts among Snake River stocks, or between upstream and downstream stocks) could improve evaluations of factors affecting populations. **The job of future analysts and decision makers will be easier if the 1999 decision is planned so as to not only meet survival and recovery goals, and socio-economic objectives, but within those constraints also maximize the amount of learning about key uncertainties.**

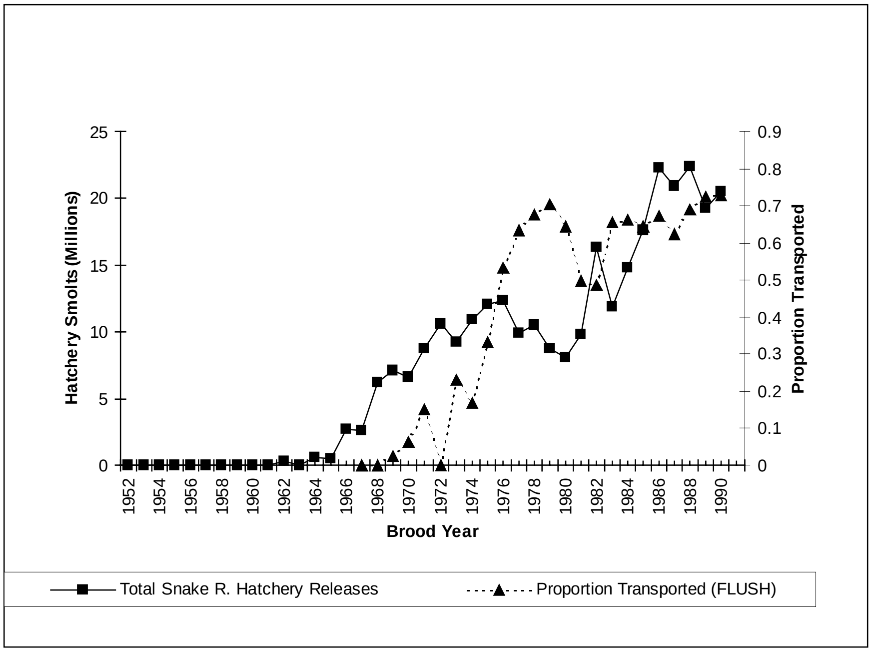

Figure 6.2-1: Confounding of hatcheries and transportation.

Experimental management provides an interface between management and science. Its allows managers to make decisions under uncertainty and use science as a tool for learning, reducing uncertainty, and improving future management decisions. This is an important point that is not often understood: experimental management and basic research are not the same; they have different purposes (Table 6.2-1). These differences mean there is a tradeoff between learning and management objectives between these two approaches (Figure 6.2-2).

Table 6.2-1: Some differences between experimental management and basic research.

	Basic Research	Experimental Management
Objective	increase understanding	increase understanding and produce goods and services
Questions addressed	mechanistic hypotheses about how nature works	what is the effect of different management actions?
Variables measured	intensive measurement of many variables (e.g., spawner abundance, water quality, habitat condition, climate)	measure only bottom line (key) management indicators (e.g., R/S, SAR)
Scale of inquiry	reductionist/small scale: often studies small areas (plots) or parts of systems	integrative/large scale: studies large areas or whole systems (cross-scale processes)
Degree of experimental control	tight control	“messy” experimental space
Results	if no difference among treatments, can’t publish	if no difference between treatments, managers happy, can use any approach, more flexibility
Who learns	scientists	managers and scientists learn
Who drives process	scientists lead, managers may support	managers lead, scientists must support (provide service)
Justification	knowledge for the sake of knowledge	must show that new knowledge will improve decision making
Utility	often idealistic	realistic

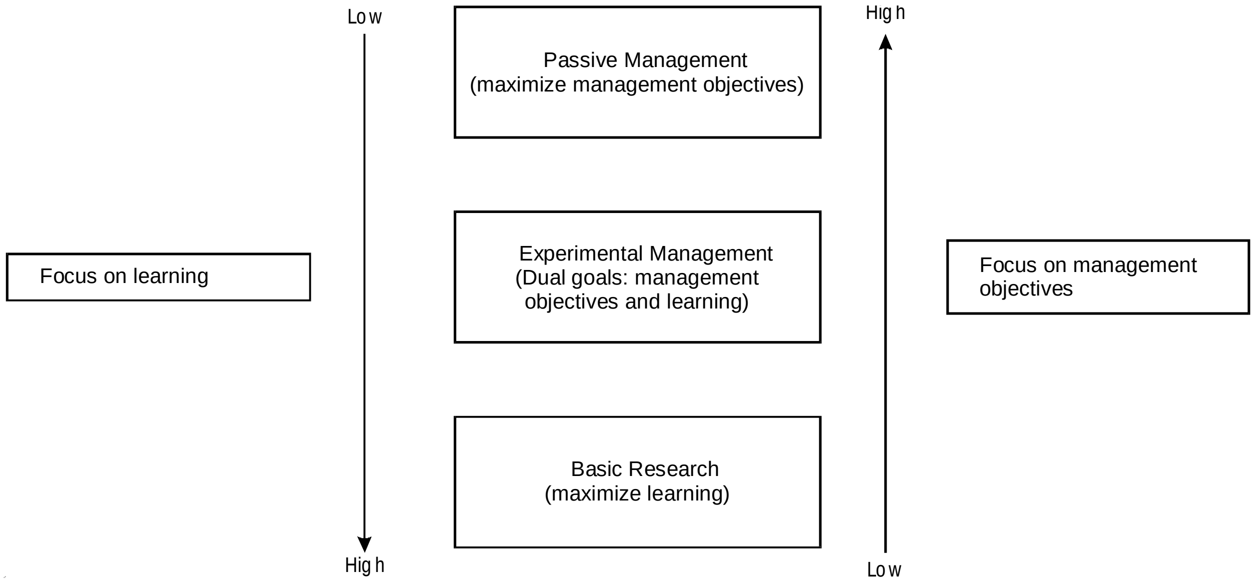

Figure 6.2-2: Tradeoff between learning and management objectives. This figure does not indicate the actual performance of a given option in meeting management objectives. It indicates the relative importance of different objectives with a given approach. Passive management does not consider learning as an explicit objective in making decisions. One can visualize a number of examples where experimental management would provide a higher ability to meet management objectives.

Experimental management proceeds in a six-step learning cycle (Table 6.2-2): assess, design, implement, monitor, evaluate, and adjust. PATH has made significant progress on Step 1. This will be important for Step 2.

Table 6.2-2: Six steps of the experimental management learning cycle.

1. Assess: determine what uncertainties to address.
2. Design: develop experimental designs that can differentiate between policies/actions.
3. Implement: conduct the experiment.
4. Monitor: monitor key management indicators over the experimental period.
5. Evaluate: compare experimental results using a pre-selected decision rule.
6. Adjust: adjust management actions based on the outcome of the evaluation process.

Assess: In this step, managers and scientists assess what key uncertainties exist and develop candidate policies that will be best for managing under a particular each set of beliefs. PATH has achieved this to a large extent through retrospective, prospective and decision analyses. PATH scientists have agreed on what data to use for analyses, developed new data, and developed state-of-the-art analytical techniques for modeling hypotheses and comparing management actions. Their analyses have defined seven key management uncertainties that most affect the survival and recovery of Snake River anadromous salmonids under a set of potential management actions (Table 6.2-3). Using this information, PATH can explore experimental management actions that can help resolve or reduce these key management uncertainties for improved future decisions.

For example, one key uncertainty is the passage and transportation assumptions incorporated in passage models (Table 6.2-3). The passage models incorporate different hypotheses about the relative benefits of transportation and flow on the survival of smolts to below Bonneville dam. These models, together with the results of transportation experiments, are used to calculate the ratio of post-Bonneville survival for transported and in-river fish (*D*), which has an important influence on the forecasted outcomes of management actions. There are however no direct empirical estimates of *D*. Thus, management experiments that provide information on the relative survival of transported and in-river fish after they leave the hydrosystem (and what hydrosystem factors influence this survival) would be valuable for reducing the uncertainty about the effects of alternative management actions. We present examples of experimental options in Section 6.3.

Table 6.2-3: Seven key uncertainties from the PATH Weight of Evidence Report.

Passage/transportation assumptions
Extra mortality
Life-cycle model
Length of transition period to equilibrium conditions after dam drawdown
Historical turbine/bypass mortality
Predator removal effectiveness
Juvenile survival rate once river has reached equilibrium conditions after drawdown

The assessment process is iterative. PATH will require feedback and participation from the Implementation Team, input and guidance from the SRP, and information from other regional management groups (e.g., hatchery and harvest managers).

Design: The design phase seeks to determine, for any given management action under consideration, what form of implementation would maximize learning while at the same time meeting management objectives. The design process considers management objectives, alternative hypotheses, management indicators, and the actual statistical methods that will be used to analyze future data. The statistical methods include the spatial and temporal pattern of data sampling. PATH should evaluate different experimental management designs prior to their implementation (Step 3). The ideal experimental design would provide the highest probability of meeting survival and recovery objectives and most quickly have the highest probability of detecting a true effect, while reducing confounding with other effects. This ideal will need to be tempered by other management objectives (i.e. the tradeoff in Figure 6.2-2). The PATH experimental evaluation process is discussed in more detail in Section 6.4.

The final four steps of the experimental management cycle (implement, monitor, evaluate, adjust) lie in the future after a decision has been made on how to proceed. However, Steps 3-5 will be important to simulate in the design phase.

While we focus on actions or combinations of actions in this document, note that continuation of the status quo is also an experiment, but an experiment not likely to generate useful information. However, learning is possible with the current hydro actions under consideration (A2, A2', A3, B1) if they are properly implemented and adequately monitored. Different implementation and monitoring plans will have to be evaluated to see how much can be learned with each action.

We note that despite our preference for an active experimental approach it is not the only approach and may not be possible in all situations. For example, Carpenter (1990) describes a passive approach where spatial and temporal replications of treatments was not possible, but time series analysis was able to deduce the relative importance of different factors. These analytical approaches however require many observations, which in the case of salmon populations translates into many decades and would be inconsistent with maximizing the probabilities of achieving survival and recovery objectives.

6.3 Experimental Management Options

The purpose of experimental management is to proceed with management actions in a manner designed to maximize the rate of learning about key uncertainties, while at the same time meeting survival and recovery objectives. For example, another key uncertainty in the PATH Weight of Evidence analyses is the cause of extra mortality of non-transported fish (Table 6.2-3). Extra mortality is any mortality occuring outside of the juvenile migration corridor that remains after accounting for: 1) productivity parameters in spawner-recruit relationships; 2) estimates of direct mortality within the migration corridor (from passage models); or 3) for the delta model only, common year effects affecting both Snake River and Lower Columbia River stocks. Extra mortality can in theory occur either before or after the hydropower migration corridor. The three alternative hypotheses explored in the PATH Weight of Evidence Report are: Hydro ("here to stay until dams are removed"), BKD ("Here to stay"), and Regime Shift ("Recurring cycles in ocean conditions"). The BKD hypothesis was used to consider any factor that would continue to cause extra mortality after changes to the hydrosystem, or changes in climate (e.g., disease, hatchery impacts) (SRP 1998). Both the SRP and the PATH Planning Group have considered example experimental management actions that could be used to differentiate between these hypotheses and thus reduce uncertainty about the causes of extra mortality.

In the Weight of Evidence Workshop, the SRP considered three experimental actions that could differentiate between hypotheses about the cause of extra mortality: transportation of smolts, operation of hatcheries, and dam drawdown. For these and other experimental management options, the SRP outlined two strategic alternatives meant to **bound** the range of feasible experimental alternatives.

1. *Incremental alternative (Figure 6.3-1a).* Implement the cheapest action first and monitor effects, then progressively more costly ones. Alternatively, if there is consensus on which actions are more likely to lead to stock recovery, those actions could be implemented first. This strategy requires the lowest up-front costs. However, it is the most risk-prone of the two strategies, because more effective actions are delayed when the cheapest action fails to produce the desired response in fish stocks. It may also be the higher-cost option in the long-term, if mitigative actions are required as each treatment is assessed. The SRP felt that the incremental approach generally characterizes past management in the Columbia River.

"Reverse staircase" or "most risk averse" alternative (Figure 6.3.1-b). This action recognizes the precarious condition of the stocks. It consists of implementing all actions at once (e.g., everything you can do to restore stocks), then turning these actions back on if appropriate after the stocks have recovered (based on an assessment of risks and benefits). This is a more risk-averse approach than the incremental approach and is more likely to lead to stock recovery, but involves larger up-front costs. Long-term costs depend on the costs and benefits associated with reversing the experimental actions (e.g., increasing hatchery supplementation operations for other threatened stocks, increasing harvest).

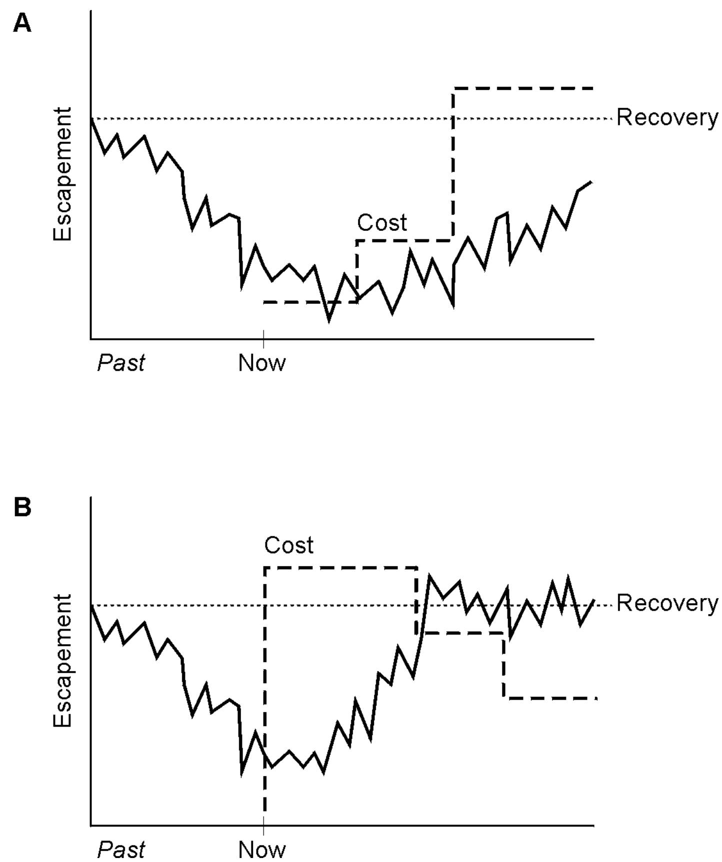

Figure 6.3-1: Alternative approaches to experimental management: A) Incremental approach; B) Reverse staircase approach.

These strategic options are meant to **bound** the range of possible experimental actions. Other possible experimental actions may include: changing the timing of delivery of transported smolts to the estuary, year to year variation in transportation or hatchery releases, or staggered / phased-in drawdown of Snake River dams. While existing transport:control studies provide information on the magnitude of transportation benefits in different flow years and for different transport groups, it is difficult to gain information about the interactive effects of transportation and hatcheries without varying both of these factors. A well designed evaluation using spatial contrast in stocks (e.g. upstream versus downstream comparisons) could enhance the ability to differentiate among hypotheses of extra mortality. Table 6.3-1 outlines a selection of example experimental manipulations associated with some of the key management actions under consideration. Although not included in this table, flow augmentation volumes are also a possible on/off experimental action (e.g., A3 vs. A5, B1 vs. B2).

Table 6.3-1: Selected examples of possible experimental manipulations associated with different decision options for spring/summer chinook, to be evaluated by PATH. Additional work needs to be completed in defining possible manipulations. Transportation: Intensive = maximize transportation; Reduce = reduce the number of fish transported or halt transportation altogether. Hatcheries: Intensive = hatcheries continue to operate as normal; Reduced = reduce hatchery production of steelhead and /or chinook in Snake River and /or the larger region. Intensive/Reduced = alternate between these treatments.

Primary Decision	**Transportation**	**Hatcheries**	**What can be learned**
Drawdown			Learn about changes in survival over time with and without dams.
4 pool drawdown (A3)	Reduced	Intensive/reduced	Can experimentally assess the impact of hatcheries on migration corridor and estuarine survival.
	Reduced	Intensive	Limits evaluation of hatcheries to passive adaptive approach.
(Most risk averse)	Reduced	Reduced	Prioritizes evaluation of hatcheries.
2 pool drawdown (2 stage implementation of 4 pool drawdown)	Intensive/reduced	Intensive/reduced	Experimentally assess the interactive effects of transportation and hatcheries on survival, while also assessing benefits of 2-pool drawdown
	Intensive	Intensive/reduced	Assumes transportation can be assessed by transport:control studies; assesses hatcheries by pulsed changes.
	Intensive/reduced	Intensive	Assumes hatchery impacts acceptable; varies transportation
	Intensive	Intensive	Assumes both hatchery and transportation impacts can be evaluated through research or continued monitoring without experimental manipulation.
(Most risk averse)	Intensive/reduced	Reduced	Assumes hatchery impacts unacceptable; varies transportation.
	Intensive	Reduced	Assumes hatchery impacts unacceptable, and that transportation can be assessed by T:C studies.
A2 Maximize Transportation			Won't learn about drawdown but assumes that T/C studies can tell you enough about benefits.
	Intensive/reduced	Intensive/reduced	Experimentally assess the interactive effects of transportation and hatcheries on survival.
	Intensive	Intensive/reduced	

6.3.1 Assessment Tasks

These examples provide some idea of what a PATH experimental management option might be like, but does not describe the specific tasks to be done in FY99. PATH must first complete the assessment step of the experimental management cycle (Section 6.2) to decide what types of experimental manipulations are worth doing, in conjunction with which hydrosystem management actions. This iterative process will require three stages: 1) define what it is we want to learn; 2) assess what we can be learn from continued monitoring, retrospective analyses, or research; and 3) contrast this with what can be learned using an active experimental management approach (which includes deliberate manipulations of management actions and an associated enhanced monitoring program). That is, PATH needs to be clear why reducing uncertainty about a particular hypothesis requires an experimental management approach, why a further perturbation is required, and why more passive approaches to learning are not adequate.

There are three important considerations for both the initial development of experimental options and the formal evaluation of those options: tradeoffs between learning and conservation; detecting results; and consultation. Each of these are discussed below.

Tradeoffs Between Experimental and Non-Experimental Actions

The region and PATH must work together to consider the relative benefits of experimental and non-experimental actions for learning and conservation, and also consider what experimental design is appropriate and/or legal under the requirements of the *Endangered Species Act*. The most informative management experiments are those that will provide the greatest contrast between treatments. However, some experiments might also increase the risk of extinction for stocks already at high risk. Thus, PATH must consider tradeoffs between learning and conservation (the probability of recovery and survival). These tradeoffs may vary between stocks, races, and species (e.g., spring/summer and fall chinook, steelhead, and sockeye salmon) and among regions (e.g., Snake River vs. Mid-Columbia). For example, increasing flows to aid spring/summer chinook smolt migration might not leave enough water to provide adequate flows for fall chinook migration.

Detecting Results

How long will it take to determine if a decision has had the desired benefits? The monitoring program used to track management indicators is a crucial component of experimental management. What variables are monitored, the frequency at which they are sampled, and the spatial and temporal resolution of data collection determines what size of effect managers will be able to detect, and how long it will take to detect it. Thus when evaluating any option, PATH must estimate how long it will take to observe responses if they exist. Larger perturbations mean less time to observe effects, but in some cases these may also increase risk to stocks. Section 6.4 discusses how monitoring will be considered in the evaluation process.

Consultation

The selection of experimental management options is an iterative process and will require input from many people. First, PATH's initial recommendations (based on the preliminary assessments of the best ways to reduce key uncertainties under each hydrosystem options) need to be reviewed and filtered by the Regional Forum Implementation Team and other decision-making groups such as the Northwest Power Planning Council. Second, the Drawdown Regional Economic Workgroup (DREW) may wish to explore the economic costs and benefits of particular experimental management options. Third, PATH cannot deal with the logistic complexities associated with changes that may be required in specific management sectors of the Columbia Basin ecosystem. When PATH has developed a set of potential experimental actions, we must consult with managers in these areas to refine and plan these actions. For example, if PATH decides that changing hatcheries or harvest policies is an acceptable experimental management option (from a conservation and learning perspective), then hatchery operators and harvest managers must be involved in the detailed planning of how to do this. This consultation will also be important to recognize and reduce the possibility of confounding between PATH experimental management options and other management activities taking place in the Columbia River basin.

6.3.2 Design Tasks

As part of the design process for a management experiment, PATH should explicitly state:

- which hypotheses are to be tested (hypotheses must be clearly stated and quantitative);
- the specific management actions that would be taken to test these hypotheses;
- the sequence and schedule (both within and between years) of treatments;
- the spatial scale of actions;

- the indicators that would be monitored;
- how indicators would be monitored (i.e., existing data collection system vs. new system);
- the frequency, timing, and duration of monitoring;
- how the information gathered would be used to evaluate hypotheses; and
- what responses would be expected if each of the hypotheses were correct.

6.4 Evaluation of Experimental Management Options

The evaluation of experimental management options falls within the design step of the experimental management cycle (Table 6.2-2). PATH would assess the quantitative performance of each option with respect to both conservation objectives (e.g., probability of survival and recovery) and monitoring objectives (e.g., ability to detecting an effect). The evaluation of experimental management options has been addressed by the SRP. This section summarizes their advice.

In their recent report, the SRP discussed what could be done to evaluate experimental actions and provided PATH with some guidance for this task (SRP 1998). Their advice touched on the following key points.

- **Basic approach:** An experimental management action is evaluated by assuming some underlying hypothesis that relates system response to that action, then simulating the effects of that action. The simulated data include both sampling error and natural variability. The data are analyzed to see if the hypothesized response can be detected and how long it will take to detect. This approach is known as “gaming”; a good description is found in Walters (1994).
- **Develop simple evaluation tools:** The current PATH passage and life cycle models are complex, too time-consuming, and inflexible for exploring experimental management options. For rapid evaluation of experimental designs, PATH should develop a simpler set of tools (hypotheses and models). Key features of these simpler tools are:
  - they must be able to capture the key dynamics and uncertainty that is observed in the more detailed and complicated models and allow rapid evaluation of alternative experimental designs.
  - they need only predict recruitment anomalies in response to the experimental actions under different hypotheses about the response of R/S to the experimental treatment.
  - they should remove the effects of other factors on R/S (e.g., density-dependent spawner effects, common year effects, and the main in-river survival effects of lower dams).
  - they will require some form of passage model to remove the main in-river survival effects of lower dams. A simplified version of FLUSH or CRiSP has already been proposed by Dr. Jim Anderson; and other simpler models have been explored by Dr. Rick Deriso (Chapter 5 of PATH FY96 Retrospective Report (Marmorek 1996)).
  - The SRP emphasized that **using simpler models does not mean throwing the detailed models away.** The simpler models must be checked for consistency against the detailed passage models. For example, the primary models could generate data based on a set of assumptions under a specific hydrosystem management action. These data would capture the variability in key indicators expected under those conditions. Simple relationships between relevant management variables and key indicators could

be developed using this data. These relationships could be included in simpler models that would be able to generate data in response to simulated management experiments.

- **New hypotheses:** New hypotheses have been introduced that have not yet been explicitly modeled. For example, the SRP found that a hypothesis about the impact of hatchery releases on extra mortality was credible and worth further consideration (SRP 1998). To evaluate this hypothesis, transport:control ratios should be monitored to learn about the differential effects of hatcheries on transported vs. non-transported fish and compare SARs of wild lower river stocks to transport and control SARs.
- **Key management variables to monitor:** The change in recruitment anomalies (i.e., R/S) is the crucial management observation required to distinguish between hypotheses. PATH needs to look at mortality over the entire life cycle because of the potential for compensatory/ depensatory mortality in individual life stages. Thus evaluation should focus on the expected response of R/S to an experimental action. However, information on SAR and reach survival would be useful as well.
- **Analytical framework for evaluation:** decision analysis can be used to evaluate alternative experimental designs for implementing a given management action, as well for comparing alternative management actions.

The schematic diagram in Figure 6.4-1 captures the relationship between the primary PATH models and the simpler models used in the evaluation process. This diagram can be broken into three components, each of which provides a different set of performance measures to use in the evaluation process. First, a simpler set of simulation models produce data with natural variability. The observation model to assesses the ability of monitoring to detect true management effects. This model adds sampling error to the data. The data are then analyzed using the same methods that would be used in actual analyses of experimental results. Both natural variability and sampling error will affect the ability of the experiment to detect changes that result from management actions. This phase produces monitoring performance measures (e.g. probability of detecting effects). The bottom box in Figure 6.4-1 concerns the valuation of experimental management designs based on economic performance measures. This is not part of the PATH process. It may be something for DREW to consider in the future. These performance measures could also be used in the assessment of tradeoffs. The models can be used to simulate the impact of other actions (e.g., harvest, hatchery, habitat), providing defensible hypotheses can be developed to link these actions to changes in indicator variables. It will be important to constrain the number of hypotheses. Only a limited number of hypotheses can be considered in the time available.

Conservation and monitoring performance objectives can be used to explore tradeoffs between management and learning. For each experimental design and underlying set of hypotheses linking actions to effects (using simpler models), PATH would estimate the probability of survival and recovery and the probability of detecting a true effect of a desired magnitude. For example, Table 6.4.1 illustrates how this tradeoff might occur for an evaluation of four hypothetical experimental options. We emphasize that this example is hypothetical and is not meant to imply what PATH will find. One design (Exp1) shows a high probability of meeting jeopardy standards, but the monitoring analyses show that it has a poor ability to generate detectable effects. A second design (Exp4) meets the conservation standards equally as well as Exp1, but it has a higher ability to detect experimental effects. Exp2 has high ability to detect effects, but does not meet conservation objectives as well as Exp1 or Exp4. Exp3 is rated last for all three categories.

Table 6.4-1: Rating of how well four designs meet management and learning objectives: 1 = high, 2 = medium, 3 = low

	Management		Learning
Experimental Design	**Pr(recovery)**	**Pr(survival)**	**Ability to detect effects**
Exp1	1	1	3
Exp2	2	2	1
Exp3	3	3	3
Exp4	1	1	1

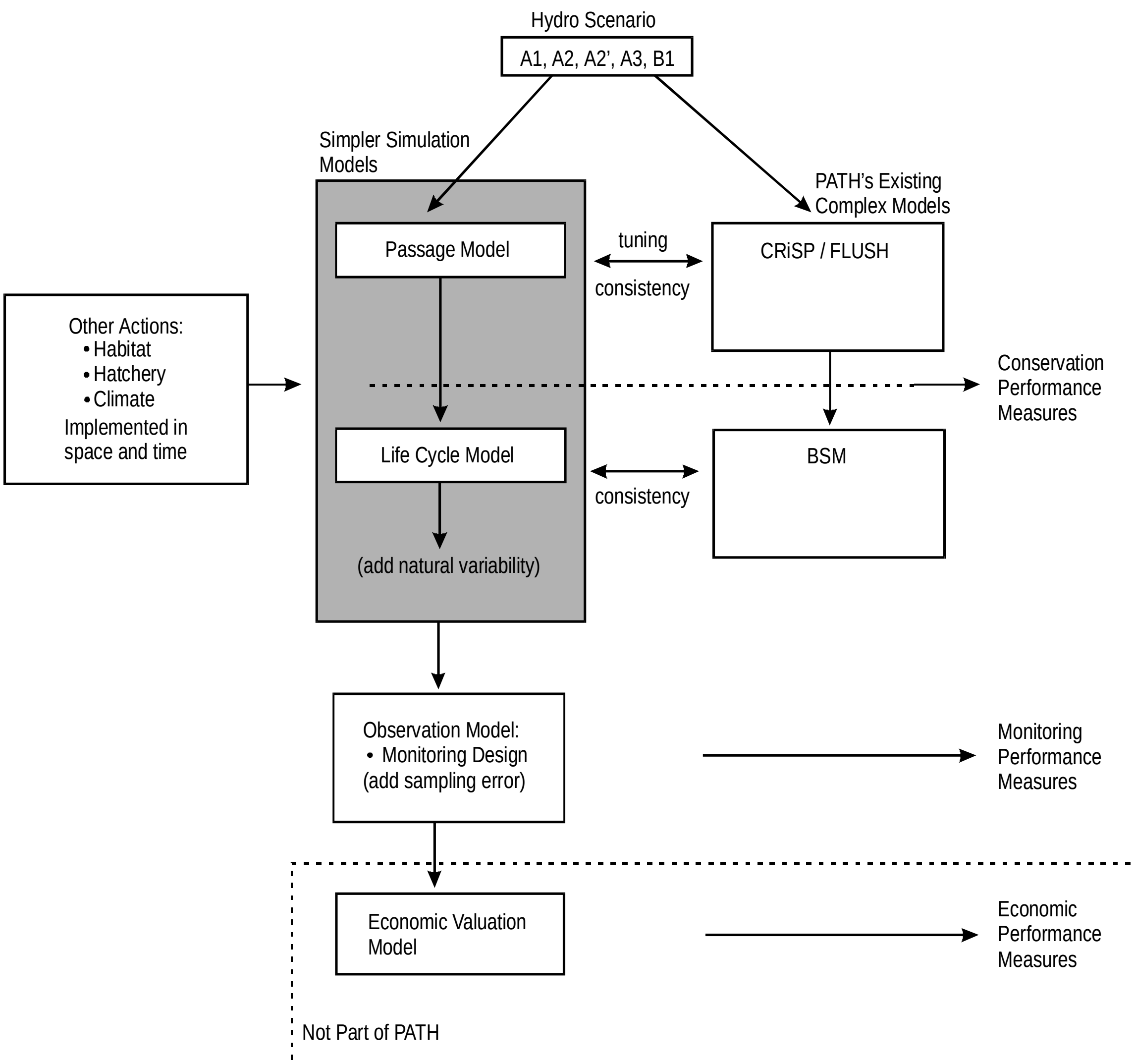

Figure 6.4-1: Schematic relationship of existing complex models to simpler models used to evaluate experimental management options, and evaluation performance measures.

6.5 Experimental Management Objectives for FY99

Table 6.5-1 summarizes the basic objectives for PATH FY99 work in experimental management. In the context of the adaptive-management cycle, the first four objectives in Table 6.2-2 cover the assessment step and initial phase of the design step. EM5 and EM6 complete the design step.

Table 6.5-1: Basic objectives of PATH FY99 work in experimental management:

EM1	Clarify the EM approach the SRP has recommended.
EM2	Describe the EM options as variations to A1, A2, A3, etc.: review by the Implementation Team.
EM3	More detailed description of EM options with review, input from SRP.
EM4	Develop tools (modifying models, developing simpler models, compare simpler models to existing ones) for quickly evaluating EM options.
EM5	Evaluate proposed management actions with/without EM options in terms of risk to stocks versus amount of learning possible.
EM6	Evaluate proposed management actions with/without EM options across populations (e.g., spring/summer and fall chinook).
EM7	Using results from EM evaluation, develop a research, monitoring, and evaluation plan to support the 1999 decision.

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7.0 PATH Analyses in FY99

Throughout this report, we have highlighted additional analyses that PATH could undertake in the next Fiscal Year. These include further sensitivity and diagnostic analyses of spring/summer and fall chinook, development of quantitative analyses for steelhead and sockeye, and further development and evaluation of experimental management options. In addition to these analyses, PATH may be called on to participate in analyses of mid-Columbia steelhead and spring chinook in support of future NMFS Biological Opinions. PATH's role in these analyses is presently unclear, but could range from an advisory/review capacity to acting as the lead analytical group. Over and above the analytical tasks of PATH are the coordination activities and presentations through which PATH interacts with policy making groups (possibly including the regional framework process), and the public.

The following table (Table 7-1) summarizes all of the tasks that have been proposed for PATH for Fiscal Year 1999. Because of time and personnel constraints, PATH will be unable to complete all of the proposed tasks listed in Table 7-1. PATH, therefore, is currently working with the Regional Implementation Team to prioritize these tasks and develop a PATH workplan for FY99. The workplan will consist of a subset of the tasks listed in Table 7-1.

Table 7-1: Tasks proposed for PATH in FY99. PATH is currently working with the Regional Implementation Team to prioritize these tasks.

Month	Experimental Management	Snake R. Spring/Summer	Snake R. Fall	Snake R. Steelhead	Mid-Columbia Stocks	PATH Coordination / Presentations
Nov. 98	Describe Exp. Mgmt Options (Note 1); Review by I.T.; Short report on proposed methods for EM	Complete PATH FY98 Final Report (Note 2)			Initial exploration of PATH role in Mid-Columbia analyses (Note 3)	FY99 workplans, schedule, and priorities; Review by I.T. (Note 4)
PATH FY98 FINAL REPORT; NMFS BIOLOGICAL APPENDIX; FY99 Work Plans						
Dec. 98	SRP Review of PATH Products				Analyses of mid-Columbia stocks	Presentation to IT, EC
Jan. 99	Further definition of options; Modify and test simplified models/tools; Evaluate Exp. Mgmt. options (Note 5)	Further sensitivity analyses and possible modeling analysis of A6/A6' (Note 6)	Further prospective analyses and assessment of options, possibly including A6 (Note 7); Weight of Evidence process for key hypotheses	Further qualitative prospective analyses		
Feb. 99						
	PATH Workshop to review results and write Update to FY98 Report					
Mar. 99	Write / review Update to FY98 Report					Presentations to NPPC, IT, ISAB, public
NMFS INPUT TO DRAFT FEASIBILITY STUDY; UPDATE TO PATH FY98 FINAL REPORT (Note 8)						
Apr. 99	Draft Report on Experimental Management Options	SRP Review of Update to FY98 Report			Analyses of mid-Columbia stocks;	Input to Multi-Species Framework (Note 9)
May 99		Publish retrospective & prospective analyses on Snake River sp/sum and fall chinook and steelhead				
June 99						
July 99						
Aug. 99	SRP Review; Modifications	Write / review Final Report for FY99				
Sept. 99						
Oct. 99	Final Report for FY99 (incl. Research, monitoring, and evaluation plans for 1999 decision)					Work Plan for FY2000; Presentations of results
FINAL REPORT FOR FY99						

Notes:

1. Initial description of experimental management options will be at a general strategic level, with examples.

2. The PATH FY98 Final Report will include the following:
   - decision analyses of actions A1, A2, A2', A3, and B1 for spring/summer & fall chinook (preliminary analyses for fall chinook);
   - Sensitivity analysis of effects of habitat (spring/summer chinook) and harvest (spring/summer chinook and fall chinook)
   - preliminary, comparative analysis of performance of A6/A6' relative to other actions;
   - qualitative assessment of steelhead, based on comparison to spring/summer chinook;
   - qualitative discussion of sockeye; and
   - brief description of exp. management – purpose, options, process for evaluating options.

3. It is not clear at this point how much work PATH will do on mid-Columbia stocks. The tasks could range from merely providing advice and review to others outside of PATH who are doing detailed analyses (a modest amount of time), to a major effort, involving: retrospective analyses; developing or adapting analytical tools; prospective analyses; and applying decision analyses to these stocks. The latter option would make it impossible to complete the other tasks shown within the time indicated.

4. PATH will need I.T. subgroup's input on relative priority of tasks during Dec./ 98 - Feb/99 (modeling of A6/A6' vs. further sensitivity analyses vs. experimental management vs. mid-Columbia analyses) and extent of treatment of other H's.

5. Experimental management options will be defined in more detail at this stage, with input from the SRP. Options will be evaluated using modified models/tools. Performance measures are benefits and risks to stocks vs. amount of learning possible.

6. Further sensitivity analyses for spring/summer chinook could include:
   - improvements in dam / reservoir survival needed to meet all jeopardy standards;
   - further analyses of effects of habitat (including roads and water quality issues), hatchery and harvest ;
   - effects of avian predation control;
   - issues with respect to data and drawdown assumptions; and
   - alternative methods for estimating D values.

7. Further prospective analyses for fall chinook could include:
   - further analyses of effects of modeling migration only vs. migration and rearing;
   - effects of B1 on spawning habitat and rearing productivity;
   - alternative methods for estimating D values;
   - review of survival and recovery levels; and
   - wider range of harvest rates.

8. Update will include:
   - results of further sensitivity/prospective analyses for spring/summer & fall chinook, and for steelhead;
   - possibly results of modeling analyses of A6/A6’ for spring/summer and fall chinook; and
   - preliminary evaluation of experimental management options.
9. As with the mid-Columbia work, it is not yet clear how much effort this may involve by PATH scientists.

Glossary of Terms, Acronyms, Variables and Parameters

α: extra mortality in a given year for a given sub-region (i.e., Snake River, Lower Columbia River).

δ: year – effect parameter for a given year (common year effects affecting both upstream and downstream stocks).

ε: normally distributed mixed process error and recruitment measurement, which depends on year and sub-basin.

λ: post-Bonneville survival factors for transported (λ_t) and non-transported smolts (λ_n).

μ: incremental total mortality between the Snake River Basin and the John Day project in a specific year.

ω: system survival (e^{-M} + [DP + 1 – P]).

a: Ricker *a* parameter.

b: Ricker b parameter.

A1, A2, A2', A3, B1: Management Actions (see Table 1.1-2).

Aggregate hypothesis: a set of alternative hypotheses about all components of the system (stock productivity, downstream migration, marine survival, etc.).

AIC (Akaike Information Criterion): *-2 ln (Likelihood) + 2p*, where p = #parameters.

Alpha Model: one of two models of salmon population dynamics used in the PATH prospective analyses. It is based on a Ricker stock-recruitment function, with additional terms for direct juvenile passage mortality and for remaining additional mortality from natural and anthropogenic causes. These two terms are assumed to be specific to the Snake River, Mid-Columbia, and Lower Columbia regions (see **Delta Model**).

BIC (Bayesian Information Criterion): *-2 ln (Likelihood) + p*ln(k)*, where p = # parameters and k= # observations.

BKD (Bacterial Kidney Disease): a serious salmonid disease which can cause death or health impairment in both juveniles and adults.

BOD (Bonneville Dam Observation Program: An accounting of the fall chinook at Bonneville Dam by bright (upriver late maturing stocks) and tule (lower river early maturing stocks) designation.

BON (Bonneville Dam)

BPA (Bonneville Power Administration)

BRWG (Biological Requirements Working Group)

BSM (Bayesian Simulation Model)

BY (Brood year): the year in which a fish was propagated or spawned.

Brights: late maturing fall chinook typically from above The Dalles Dam. Bright in color and not yet ready to spawn when they enter the mouth of the Columbia River.

CARTs (Categorical Regression Trees)

cp: complexity parameter.

CPUE (Catch Per Unit Effort)

CRFMP (Columbia River Fish Management Plan) an agreement between sovereigns that allocates fishing effort in accordance with a harvest schedule designed to rebuild stocks and meet treaty obligations with Native Americans.

CRiSP (Columbia River Salmon Passage Model)

CWT (Coded wire tag): a tiny tag (1 x 0.25 mm) generally imbedded in the nose cartilage of fingerling or fry while the fish is still in the hatchery. The coded tag allows detailed data on brood year, date of release, and other information to be obtained when the fish is recaptured years later.

D: ratio of post-Bonneville survival of transported fish to post-Bonneville survival of in-river fish.

Delta Model: one of two models of salmon population dynamics used in the **PATH** prospective analyses. It is based on a Ricker stock-recruitment function, with additional terms for direct juvenile passage mortality, an extra mortality factor, and a common year effect. The direct and extra mortality terms are region-specific, while the common year effect acts on all regions (see **Alpha Model**).

Depensatory: a process that causes mortality rates to increase as abundance decreases. An example of a depensatory process is when the number of individuals removed by predation remains constant as the population abundance decreases.

DES: the naturally spawning bright fall chinook index stock from the Deschutes River. A secondary component of the **URB** harvest management unit.

Drawdown: releasing water from a reservoir to lower its elevation, thereby reducing surface area and cross-section. This increases water velocity (at any given discharge) in comparison to velocities at higher water levels in the reservoir.

E: climate index variable (**PAPA** drift). Represents the latitude of a drifting object after three months drift starting at station **PAPA**.

EJUV: Equilibrated Juvenile survival rates following drawdown.

EM (Extra Mortality): extra mortality is any mortality occurring outside of the juvenile migration corridor that is not accounted for by either: 1) productivity parameters in spawner-recruit relationships; 2) estimates of direct mortality within the migration corridor (from passage models); or 3) for the delta model only, common year effects affecting both Snake River and Lower Columbia River stocks.

EMCLIM: Extra Mortality / future Climate.

ESA (Endangered Species Act)

ESBS (Extended Length Submersible Bar Screens)

ESU (Evolutionary Significant Unit): a population or group of populations that is considered distinct (and hence a "species") for purposes of conservation under the **ESA**. To qualify as an ESU, a population must: 1) be reproductively isolated from other conspecific populations; 2) represent an important component in the evolutionary legacy of the biological species.

F: average flow (in thousand cubic feet per second) at Astoria during April-June.

FGE (Fish Guidance Efficiency): the percentage of juvenile fish approaching a turbine intake that are guided into facilities designed to bypass the turbine.

FLUSH (Fish Leaving Under Several Hypotheses): a passage model developed by the State and Tribal fish agencies.

FCRPS (Federal Columbia River Power System): the major hydropower dams of the lower Snake and lower Columbia rivers.

FTT (Fish Transit Time): the time it takes smolts to travel from the head of Lower Granite pool to the Bonneville tailrace.

GBT (Gas Bubble Trauma): non-lethal or lethal effects of the growth of air bubbles in the cardiovascular systems of fish.

HAB: habitat effects.

HYSER: a U.S. Army Corps hydro-regulation model to predict monthly flows associated with a particular method of operating the hydrosystem.

HYURB: the naturally spawning **Upriver Bright** fall chinook index stock from the Hanford Reach and Yakima River area (McNary Pool). The major component of the **URB** harvest management unit.

IHR/IHB (Ice Harbor Dam)

ISAB (Independent Scientific Advisory Board): scientific body that provides independent advice and reviews to NMFS and the NPPC.

I.T. (Implementation Team): an inter-agency policy group to whom **PATH** reports.

In-river survival rate (Vn): direct survival rate of non-transported smolts. The in-river survival rate is estimated from the top of the first reservoir encountered to below Bonneville Dam.

JDA/JDD (John Day Dam)

Jeopardy standards: main performance measures used in **PATH** preliminary decision analysis to evaluate alternative management actions and assess sensitivity of outcomes to various uncertainties. The Jeopardy standards are a measure of spawning abundance relative to pre-defined thresholds that are associated with survival and recovery of endangered stocks (see **Survival standard** and **Recovery standard**).

KCFS: a unit of measure for flowing water, expressed in thousands of cubic feet per second.

LGO/LGS (Little Goose Dam)

LGR (Lower Granite Dam)

LMO/LMN (Lower Monumental Dam)

LRW (Lower River Wild): a Columbia River fall chinook harvest management unit that is composed of bright stocks below Bonnevile Dam, including the North Fork Lewis River stock.

m: total direct passage mortality rate, including both passage and extra mortality.

Δm: extra mortality rate, expressed as an instantaneous rate, which depends on year and region, and is calculated as the differences between total mortality (**m**) and passage mortality (**M**).

M: direct instantaneous passage mortality rate of juvenile fish (both transported and non-transported) from **LGR** pool to below **BON**.

MCN (McNary Dam)

MLE (Maximum Likelihood Estimate)

NFL: the naturally spawning bright fall chinook index stock from the North Fork of the Lewis River. The major component of the **LRW** harvest management unit.

NMFS (National Marine Fisheries Service)

NPPC (Northwest Power Planning Council)

Natural river drawdown: an option for implementing drawdown of dams where the reservoir is completely drained to create a free-flowing river. This is done either by removing the earthen embankments adjacent to the dam structure, or by building a channel around the dam. In either case, diversion of water around the dam structure results in loss of power-generating capability.

Natural Spawner: Adult salmon that spawn in-river as opposed to returning to artificial spawning channels and hatcheries. Their origin may be natural or hatchery.

OSCURS: an ocean circulation model.

p: depensation parameter.

P or Pbt: the proportion of juvenile fish below **BON** that were transported.

PAPA: an index of ocean currents

PATH (Plan for Analyzing and Testing Hypotheses)

PDO (Pacific Decadal Oscillation)

PIT (Passive Integrated Transponder) tags: these tags are used for identifying individual salmon for monitoring and research purposes. The miniaturized tag consists of an integrated microchip that is programmed to include specific fish information. The tag is inserted into the body cavity of the fish and decoded at selected monitoring sites.

PMOD: Passage Model.

PREM: Predator Removal effectiveness.

PRD (Priest Rapids Dam)

PRER: length of pre-removal period.

PROSP: prospective model for the distribution of extra mortality (Alpha or Delta).

PSC-CTC (Pacific Salmon Commission Chinook Technical Committee): deals with ocean salmon harvest management issues.

Productivity: natural log of the ratio of recruits to spawners for a specified time period (in the absence of density dependent mortality). Measured here as the intercept or "*a*" value from the Ricker spawner/recruit function.

R: "observed" recruitments (returning progeny) originating from a given set of natural spawners (parents). The measurement may be taken at different points, such as the spawning ground, or the mouth of the Columbia River (including or not including ocean harvest impacts). In this document, recruits include all mature (jack and adult) returns of natural origin.

Rkm (River kilometer): a measurement of river length in kilometers typically taken from the mouth of the river or tributary to the designated landmark following the course of the river.

R/S: recruits per spawner is the number of mature fish returning to the point of recruitment (R) divided by the number of spawners in the parent generation (S).

Recovery standard: the performance measure used to describe the effect of a certain hydrosystem action on the chance of a spawning stock for recovery; the fraction of simulation runs for which the average spawner abundance over the last 8 years of a 48-year simulation is greater than a specified level (different for each stream).

s: **FLUSH** variable for survival to below **BON** of control (non-transported) fish.

S: "observed" spawners (parents). In this document, jacks are not considered to contribute to spawning, so only adult spawners are counted as parents. All adults on the spawning ground, regardless of origin, are considered to be parents for the natural-origin recruits.

SAR (Smolt-to-adult return rate): survival rates of fish from the time they pass the upper-most dam as smolts to the time they return to that dam as adults.

SRB (Snake River Brights): a Columbia River fall chinook harvest management sub-unit that is part of the **URB** unit, now tracked separately due to **ESA** listing of Snake River fall chinook. The naturally spawning bright fall chinook index stock from the lower Snake River.

SRI (Survival Rate Index): the residuals from a fit of stock recruitment function to a given period of brood years. The natural log of the ratio of observed **R/S** and predicted **R/S** from a fit of observed recruitment data to the Ricker spawner/recruit function.

SRP (Scientific Review Panel)

STEP: formulated to model the effect of a 1975 (brood year) climate regime shift, which has different effects in different subregions.

STS (Standard Length Submersible Travel Screens)

Spillway crest: an option for implementing drawdown of dams where water levels in the reservoir are lowered to approximately 60-70% of the maximum level. Turbines could continue to operate under this drawdown configuration.

Survival standard: the performance measure used to describe the possibility of extinction; the fraction of time during many simulations that the spawning abundance of a stock is above a certain specified low threshold (150 or 300 spawners depending on the characteristics of the stock and the stream).

System survival: the number of in-river equivalent smolts below Bonneville Dam divided by the population at the head of the first reservoir.

T:C or T/C or TCR: the Transport : Control ratio is the ratio of transported fish survival to in-river fish survival from juveniles at the collection point to adults at the same point.

TAC (*U.S. v Oregon* Technical Advisory Committee): advises on Columbia River harvest management issues for various species including salmonids.

TDA/TDD (The Dalles Dam)

TRANS: transportation model.

TJUV: Transition period: Juvenile survival.

TURB: historical turbine / bypass survival assumptions.

Tules: early maturing fall chinook of the lower Columbia River (not found naturally above The Dalles Dam). Dark in color and ready to spawn when they enter the mouth of the Columbia River.

URB (Upriver Brights): a Columbia River fall chinook harvest management unit that includes the Hanford Reach-Yakima River stock, the Deschutes River stock, and the Snake River bright stock.

Vn: direct passage survival of in-river juvenile fish, measured from the head of **LGR** pool to the tailrace of **BON**, including reservoir and dam survival at each project.

WOE (Weight of Evidence)

WTT (Water Transit Time)

Wild Spawner: the natural spawner whose parents were of natural origin.

Appendix A: Assumptions Needed to Apply Weights to Results for A2' and B1 for Spring/Summer Chinook

A2' and John Day drawdown had not yet been modeled when the Weight of Evidence Process was completed. Consequently, alternative assumptions specific to these actions were not considered. For A2', the effects of Surface Bypass Collectors was assumed to be primarily through an improvement in FGE. We considered two alternative assumptions about FGE under A2': a lower bound equal to the pessimistic FGE applied to A2 (FGE2), and an upper bound where FGE increased by approximately 9%, based on an assumed SBC efficiency of 32%. FGE assumptions for A1, A2, and A3 were not weighted in the Weight of Evidence Process because they had very little influence on results (*PATH Weight of Evidence Report*). Figures 2.2.3-1 to 2.2.3-4 also show that the alternative values of FGE have virtually no effect on the results. Therefore, both FGE assumptions for A2' were weighted equally (0.5 on FGE2, 0.5 on FGE3) in the weighted results.

Assumptions associated with John Day drawdown were the same as those associated with Snake River drawdown – length of pre-removal period, length of transition period, and the equilibrated juvenile survival rate. Weights were assigned to the latter two for Snake River drawdown through the Weight of Evidence process, and these weights were assumed to be applicable also to John Day drawdown because the mechanisms behind the alternative assumptions were the same for both actions [see the *Preliminary Decision Analysis Report* (Marmorek and Peters 1998] for details on the mechanisms behind these assumptions). In fact, the equilibrated juvenile survival rate and transition period assumptions for Snake River and John Day drawdown were explicitly linked in the analysis (i.e., the lower equilibrated survival rate for Snake River drawdown was only run with the lower equilibrated rate for John Day, etc.; see Table 2.2.4-2).

However, applying these weights was not straight-forward because not all possible combinations of drawdown hypotheses were run for B1. Without doing more runs, applying the weights to these situations required making some assumptions. There were two situations where this was the case:

a) The lower equilibrated juvenile survival rate was run only with a set of lower-bound assumptions about FGE, historical turbine/bypass survival (TURB), and predator removal efficiency (PREM). Similarly, the higher equilibrated juvenile survival rate was run only with a set of upper-bound FGE, TURB, and PREM assumptions. This was done to reduce the number of runs to accommodate time constraints while still bracketing the range of possible outcomes.
b) For B1, the long transition period for Snake River dams was only run in conjunction with the long pre-removal period for John Day dam drawdown. The reason for this was that if a short pre-removal period for John Day drawdown had been run with a long transition period for Snake River drawdown, the transition period following drawdown of the Snake River dams would have overlapped with the transition period following drawdown of the John Day dam. Modeling this would have required some assumptions and interpolations about the interactions of the two transition periods, and we did not have time to discuss what those assumptions should be.

For situation a), we applied the weight of the equilibrated juvenile survival rate to the entire combination of FGE, TURB, PREM and equilibrated juvenile survival rate. This effectively overrides the weights on the FGE, TURB, and PREM assumptions with the weight placed on the equilibrated juvenile survival rate. This was thought to be an acceptable resolution because of the minor effects the FGE, TURB, and PREM assumptions have on determining the modeling results (Figures 2.2.3-1 to 2.2.3-4). However, to explore the effects of this assumption, we compared weighted averages of the survival and recovery probabilities for A3 using a full set of weights, to weighted averages based on only the best/worst combination of FGE, TURB, PREM, and equilibrated juvenile survival assumptions, with the weights on the equilibrated juvenile survival rate applied to the entire combination. The best/worst weighted average is analogous to the approach applied to B1. Hypotheses other than FGE, TURB, PREM, and equilibrated juvenile survival rate were either weighted equally or were weighted with the SRP weights. The two weighted average jeopardy probabilities differed only slightly from each other. If one assumes that the response of A3 and B1 are similar (which the CART diagrams suggest), than the weighting approach applied to the FGE, TURB, and PREM assumptions in B1 does not introduce significant bias into the results.

Table A-1: Effects of full weights vs. weighting treatment of best/worst case passage and equilibrated juvenile survival rate on weighted average survival and recovery probabilities.

Weights	Full weights - A3			Best/Worst - A3		
	24-Yr Survival	100-Yr Survival	48-Yr Recovery	24-Yr Survival	100-Yr Survival	48-Yr Recovery
Equal	0.71	0.89	0.82	0.71	0.88	0.81
JC	0.75	0.91	0.91	0.75	0.91	0.90
SS	0.66	0.88	0.78	0.67	0.87	0.77
SC	0.71	0.90	0.91	0.71	0.89	0.89
CW	0.72	0.89	0.88	0.74	0.90	0.90

For situation b), we first explored the relative sensitivity of B1 results to the limited set of combinations of drawdown assumptions that were modeled. These combinations and their weighted average jeopardy probabilities are listed in Table A-2. Only the results with equal weights applied to the other (non-drawdown) hypotheses are shown here; results using the SRP sets of weights are similar.

Table A-2: Combinations of drawdown assumptions modeled for B1

	Snake R. dam Pre-Removal	Snake R. dam equil. juv. surv. rate	Snake R. dam Transition	John Day dam Pre-removal	John Day equil. juv. surv. rate	John Day Transition	24-yr Surv.	100-yr Surv.	48-yr Rec.
1	3 years	Low	2 years	10 years	Low	2 years	0.71	0.89	0.83
2	3 years	Low	2 years	15 years	Low	2 years	0.72	0.89	0.83
3	8 years	Low	2 years	10 years	Low	2 years	0.69	0.88	0.81
4	8 years	Low	2 years	15 years	Low	2 years	0.68	0.88	0.81
5	3 years	Low	10 years	15 years	Low	10 years	0.68	0.88	0.81
6	8 years	Low	10 years	15 years	Low	10 years	0.66	0.87	0.81
7	3 years	High	2 years	10 years	High	2 years	0.77	0.91	0.88
8	3 years	High	2 years	15 years	High	2 years	0.77	0.91	0.90
9	8 years	High	2 years	10 years	High	2 years	0.73	0.91	0.89
10	8 years	High	2 years	15 years	High	2 years	0.73	0.91	0.90
11	3 years	High	10 years	15 years	High	10 years	0.75	0.91	0.89
12	8 years	High	10 years	15 years	High	10 years	0.72	0.90	0.89

Weighted average jeopardy probabilities varied by up to 0.10 between sets of drawdown assumptions. The most influential factor appeared to be the equilibrated juvenile survival rate – results using the high (Scenarios 7-12) and low (1-6) equilibrated juvenile survival rate tended to produce relatively homogenous groupings of results. Within each of these sub-groupings, results tended to vary little and there was no consistent patterns in the assumptions. These results are consistent with the relative influence of hypotheses in the CART diagrams.

With regard to weighting, the results suggest that it is important to weight the equilibrated juvenile survival rates. We were able to do this because there are an equal number of runs with both hypotheses about equilibrated juvenile survival rate. It is also possible to weight the pre-removal period for Snake R. dams because of the balanced set of runs. In previous presentations of A3 results we have not weighted the pre-removal hypotheses but have shown them separately to explicitly show the implications of a delay in any drawdown action. However, showing the pre-removal effects separately for B1 would require four different sets of results for B1 (long/short delay for Snake R. drawdown X long/short delay for John Day drawdown). Therefore, to simplify the presentation we weight the Snake R. pre-removal periods equally and show only a single set of results for B1.

Applying the weights to the three scenarios involving the John Day pre-removal and the transition period hypotheses is not possible without making certain assumptions, but these assumptions will not have a large effect on the results, based on the results in Figures 2.2.3-1 to 2.2.3-4 and Table 2.2.4-3. There are three combinations of these two uncertainties that were modeled:

A. Short transition period for Snake R. and John Day drawdown X short John Day pre-removal

B. Short transition period for Snake R. and John Day drawdown X long John Day pre-removal

C. Long transition period for Snake R. and John Day drawdown X long John Day pre-removal

We used the following approach to apply weights to the results of these three scenarios. First, we applied equal weights to A and B and calculated a weighted average (AB). Then, we applied the SRP (or equal) weight assigned to the short transition period to AB, and the SRP (or equal) weight assigned to the long transition period to C, and calculated a weighted average of AB and C.

The results presented in Section 2.2.4 use this approach for weighting B1 results. This approach assumes that the results with a short B1 pre-removal period and a long transition period, if modeled, would be similar. Again, even if this assumption were not correct, the effects on the overall weighted results for B1 would be very small.

In conclusion, we are reasonably confident that although the runs we completed for B1 are fewer than A3, and unbalanced, this has a negligible effect on the results.

Appendix B: Smolt to Adult Return Rate Estimates of Snake River Aggregate Wild and Hatchery Steelhead

Snake River wild summer steelhead populations declined with completion of the Federal Columbia River Power System (FCRPS), from run sizes of 42,000 to 106,000 (at the uppermost dam) during the 1960s to run sizes of 7,000 to 19,000 during 1990-1995 (WDFW and ODFW 1996; Hassemer et al. 1997). Before the FCRPS was completed, the Snake River produced 55% of the summer steelhead entering the Columbia River (Mallet 1974). Snake River steelhead were listed as "threatened" under the Endangered Species Act in 1997. Under the ESA, the National Marine Fisheries Service (NMFS) is charged with developing and implementing management plans to ensure survival and recovery of the listed anadromous populations.

Following ESA listings of Snake River sockeye, spring/summer chinook and fall chinook populations in 1992, the NMFS 1995-1998 Biological Opinion on operation of the FCRPS (NMFS 1995) created a process called PATH – Plan for Analyzing and Testing Hypotheses.

PATH analyses of spring/summer chinook have focused on temporal/spatial responses in productivity and survival rates based on spawner-recruit data at the individual population level (Schaller et al. 1996; Deriso et al. 1996). There is a paucity of spawner-recruit data for individual steelhead populations, however. This is due to the species' complex life cycle, spawn timing and difficulty of monitoring redds, limited ability to recover kelts for age composition estimates, the logistics and cost of weir operation, and funding processes which have favored chinook salmon research.

Raymond's (1988) estimates of smolt-to-adult return rates (SAR) for Snake River wild and hatchery steelhead for the 1964-1984 smolt migrations provide a basis to compare retrospectively changes in productivity and survival rate before, during and after FCRPS construction. Raymond's estimates and the updated data set should also be helpful in establishing survival and recovery SAR objectives for Snake River steelhead (e.g., Toole et al. 1996).

NMFS (1998) compared the ratio of wild spring/summer chinook SARs in a recent period (smolt years 1992-1994) and SARs in a historical period (smolt years 1964-1969) when Snake River spring/summer chinook were considered to be healthy. The ratio of SARs between periods for spring/summer chinook was then contrasted with that for wild steelhead. NMFS (1998) used this comparison to make inferences about the relative health of the two species, and the potential need for additional management measures for steelhead.

This progress report presents updated SAR estimates for wild and hatchery Snake River steelhead for smolt migration years 1985-1994, and compares SARs of wild Snake River steelhead with those of spring/summer chinook (Petrosky and Schaller 1998). Steelhead SAR estimates differ slightly from previous estimates (Petrosky 1998) because of FGE assumptions and refinements in estimating age-structured recruits. Two measures of SAR were defined for this analysis, consistent with the definitions in the NMFS (1998) Supplemental Biological Opinion. SARs were calculated on the basis of adult recruits to the uppermost dam (SAR1) and total adult recruits, including mainstem Columbia River harvest (SAR2).

B.1 Snake River Steelhead Runs

Snake River wild and hatchery steelhead have A-run and B-run components which are differentiated by production area, adult entry timing into the Columbia River, stock-specific differences in size and age, and differences in lower Columbia River harvest rates. In the Columbia River Basin, B-run steelhead are unique to the Clearwater River (including Lochsa, Selway and South Fork Clearwater rivers), the South Fork Salmon River and Middle Fork Salmon River, Idaho (Kiefer et al. 1992). The B-run habitat represents a majority (53%) of available steelhead habitat within Idaho (S. Kiefer, IDFG memo 3/13/98 to P. Dygert, NMFS). Hatchery B-run steelhead were derived from the North Fork Clearwater River stock, which was blocked from natal habitat by construction of Dworshak Dam in 1969. Compared to A-run steelhead, B-run fish characteristically tend to have an older ocean-age composition, as well as a larger size at ocean age. With some exceptions, B-run steelhead adults return later (generally after August 25) to the mouth of the Columbia River (Kiefer et al. 1992; TAC 1997). Recent Columbia River harvest rates (1985-1995) for wild A-run and wild B-run steelhead have ranged from 11% to 25% and from 24% to 47%, respectively (TAC 1997; "length method").

B.2 Methods

Two measures of SAR were defined for this analysis, consistent with the definitions in the NMFS 1998 Supplemental Biological Opinion. SAR1 is defined as number of adults returning to the uppermost dam from a smolt migration year divided by the total smolt outmigration. SAR2 represents the total adult recruits (catch plus escapement) from a smolt migration year divided by the total smolt outmigration, as defined by Raymond (1988). SAR2 (catch + escapement) is most relevant to evaluate the effects of FCRPS development and operation since it accounts for different harvest rates over the historic period. SAR1 (escapement) seems the more relevant to ESA survival and recovery standards, which address all mortality sources, including harvest. Note that neither SAR measure accounts for upstream passage mortality.

Run year is defined here as the year an adult steelhead enters the Columbia River since adults spend up to nine months migrating to the spawning grounds and pass upstream dams from fall through spring. For instance, adults entering the Columbia River in summer 1985 would pass the uppermost Snake River dam sometime between late summer 1985 to spring 1986.

Wild and hatchery smolt abundance at the uppermost dam was estimated for each of the 1985-1994 migrations to update Raymond's (1988) time series for the 1964-1984 migrations. Smolt abundance for each of the 1985-1994 migrations was determined as the smolt passage index at Lower Granite Dam (LGR) divided by an estimated fish guidance efficiency (FGE). The passage index represents a relative indicator of population abundance, computed by dividing the daily fish collection estimate by the proportion of flow passing through the sampled unit or powerhouse relative to the river flow (FPC 1994). Separate smolt passage indices are available for hatchery and wild steelhead, but they are not partitioned into A-run and B-run components. Wild and hatchery steelhead smolt passage indices for 1990-1994 were provided by the Fish Passage Center (D. Marvin, pers. comm.). Wild and hatchery steelhead smolt passage indices for 1985-1989 were approximated by dividing the annual collection totals (FTOT annual reports; e.g., Koski et al. 1988) by the quantity [1.0 minus proportion spill] during the spring migration season (4/15-6/15). For years with no spill, the approximation is equivalent to FPC daily summation technique. Spill proportion was nil or small in the years 1985-1989 (0.00 in 1987-1990; 0.02 in 1985; and 0.11 in 1986). FGE was assumed to be 0.74 for smolt migration years 1985-1990, and 0.81 for 1991-1994 (Krasnow 1998).

Adult returns for run years 1985-1997 of wild A-run, wild B-run, hatchery A-run and hatchery B-run steelhead were obtained from LGR return and harvest rate estimates from the U.S. vs. Oregon Technical Advisory Committee. LGR adult returns of wild A-run, wild B-run, hatchery A-run, and hatchery B-run steelhead for run years 1986-1997 were from TAC (1997, Tables 12-15), using the "length method" of run separation at LGR. Separation of A-run and B-run components is possible only by the "length method" at LGR because adult steelhead pass this point over a nine-month period. Total adult return by run year was estimated as the LGR return divided by the survival rate through the Zone 1-6 lower river fisheries (i.e., 1.0 minus harvest rate). Harvest rates for these four components of Snake River steelhead, 1985-1995, were also from TAC (1997, Tables 12-15), using the "length method" of run separation for lower river harvest. Harvest rates for 1996 and 1997 were preliminary TAC estimates (G. Mauser, IDFG, pers. comm.).

Age-structured estimates of adult return to LGR were determined from TAC estimates of annual LGR adult returns of wild A-run, wild B-run, hatchery A-run, and hatchery B-run steelhead, and from annual ocean-age composition estimates at LGR based on scale analysis from 1985-1994 run years (IDFG data; Hall-Griswold 1995). In the "length method" of partitioning A-run and B-run components, TAC (1997) determined that 14% of wild A-run steelhead are larger than 77.5-cm fork length (FL), and 36% of wild B-run steelhead are smaller than 77.5 cm FL. Similarly, 1% of hatchery A-run steelhead exceed 77.5 cm FL, and 17% of hatchery B-run steelhead are shorter than 77.5 cm FL. The first step to estimate age composition by run year and group was to sort the scale data by hatchery or wild origin, as determined by scale analysis. Next, the ocean age composition was tallied for hatchery and wild groups within the two FL classes. Finally, the TAC estimates of LGR run size were partitioned into the FL classes, each of which were then multiplied by the ocean age proportions for the respective FL class. For example, if the TAC run size of wild A-run steelhead were 1,000, then 860 would be assigned to the <77.5 cm class, and 140 to the >77.5 cm class. The 860 smaller fish would be assigned the ocean age structure from wild fish <77.5 cm, and the remaining fish would be assigned the ocean age structure from wild fish > 77.5 cm. Scale samples have not been analyzed since the 1994 run year. Estimates of ocean-age composition for run years 1995 – 1997 were based on the average proportions estimated for run years 1985-1994.

Adult recruits to the uppermost dam were estimated for wild and hatchery A-run and B-run steelhead for smolt migration years 1964-1994. For smolt years 1985-1994, LGR recruits were based on the age-structured run year estimates for each group. Raymond (1988) did not report recruits to the uppermost dam for the earlier smolt years, 1964-1984. Raymond's estimates of total recruits (escapement plus harvest) for the earlier smolt years were adjusted by the annual Zone 1-6 harvest rates, assuming an average (ocean) age structure for the aggregate run.

Total adult recruits (escapement plus harvest) were estimated for wild and hatchery A-run and B-run steelhead for smolt migration years 1964-1994. For smolt years 1985-1994, total recruits were estimated by applying annual TAC harvest rates to the age-structured LGR return. Raymond's (1988) estimates were used for smolt years 1964-1984.

Smolt-to-adult return rates were estimated for wild and hatchery Snake River steelhead, smolt migration years 1964-1994, indexed as recruits to the uppermost dam (SAR1) and as total recruits (SAR2). Adult recruits were divided by the estimated smolt abundance for each migration year to estimate SAR. Smolt yields were not partitioned into A-run and B-run components.

The SAR1 and SAR2 values used in the NMFS (1998) Supplemental Biological Opinion were updated for the new steelhead estimates, and new ratios of historic SARs to recent SARs were computed (geometric mean) for smolt years 1992-1994. SAR1 and SAR2 estimates for Snake River spring/summer chinook and ratios of historical to recent SARs were from Petrosky and Schaller (1998). Note that the "recent" time period used in this analysis differs slightly from that in NMFS (1998).

Similarity in survival rate patterns between Snake River wild steelhead and spring/summer chinook was investigated graphically and by regression. SAR2 estimates of steelhead and spring/summer chinook were plotted and regressed for smolt migration years, 1964-1994. In addition, the estimated SAR2 for both species were plotted and regressed as ln(SAR2) against water travel time for 1964-1994 smolt years. The SARs were log transformed in the regressions because residuals were expected to be log-normally distributed (Peterman 1981). Annual water travel time estimates (in days from Lewiston to Bonneville Dam) for the spring migration period (4/16-5/31) were provided by the Columbia River Inter-Tribal Fish Commission (M. Karr and E. Weber, pers. comm.).

B.3 Results

Estimates of wild smolt abundance at LGR ranged from 0.6 million to 0.9 million for wild steelhead and from 3.1 million to 8.0 million for hatchery steelhead between 1985 and 1996 (Table B.3-1). Wild smolt numbers generally decreased during the period, while hatchery smolt numbers peaked in 1993.

Table B.3-1: Smolt abundance estimates (millions) of wild and hatchery steelhead at Lower Granite Dam, 1985-1996. Proportion spill estimated for 4/15-6/15 period, FGE from Krasnow (Nov. 11, 1998). Abundance estimates based on spill adjusted collection for 1985-1989, and on FPC Passage Index for 1990-1996.

			Wild Steelhead				Hatchery Steelhead			
Year	Proportion Spill	LGR FGE	Number Collected	Adjusted Collection	FPC Index	Smolt Abundance	Number Collected	Adjusted Collection	FPC Index	Smolt Abundance
1985	0.02	0.74	0.45	0.46	na	0.63	2.23	2.28	na	3.08
1986	0.11	0.74	0.54	0.61	na	0.82	2.55	2.87	na	3.87
1987	0.00	0.74	0.55	0.55	na	0.74	2.46	2.46	na	3.33
1988	0.00	0.74	0.59	0.59	na	0.80	4.15	4.15	na	5.61
1989	0.00	0.74	0.54	0.54	na	0.74	4.70	4.70	na	6.35
1990	0.00	0.74	0.70	0.70	0.70	0.94	5.44	5.44	5.44	7.35
1991	0.00	0.81	0.63	0.63	0.63	0.78	5.66	5.68	5.68	7.02
1992	0.00	0.81	0.58	0.58	0.58	0.72	3.82	3.82	3.83	4.73
1993	0.12	0.81	0.50	0.57	0.58	0.72	5.72	6.53	6.51	8.03
1994	0.18	0.81	0.48	0.59	0.52	0.64	4.22	5.17	4.71	5.81
1995	0.12	0.81	0.41	0.47	0.49	0.60	5.50	6.24	6.27	7.74
1996	0.36	0.81	0.32	0.50	0.53	0.65	4.26	6.63	6.38	7.88

Ocean age structure of returning adults at LGR by FL class is shown in Table B.3-2. One-ocean (O1) adults generally dominated the smaller size class most years for wild and hatchery steelhead. Two-ocean fish (O2) dominated the larger size class.

Age composition at LGR of wild and hatchery A-run and B-run steelhead was estimated for run years 1985-1997 (Table B.3-3). Average age composition for run years 1985-1994 of wild A-run steelhead was 52% O1, 47% O2, and 1% O3. Wild B-run age composition averaged 28% O1, 69% O2, and 3% O3. Hatchery steelhead age composition averaged 66% O1, 34% O2, and <1% O3 for A-run, and 21% O1, 73% O2, and 6% O3 for B-run.

Table B.3-2: Number of scales by ocean age (O1 - O3) at Lower Granite Dam (LGR), run years 1985-1997, for wild and hatchery steelhead by fork length (FL) criteria. Scale samples since 1994 have not been aged.

	Scale age: Wild FL<77.5			**Scale age: Wild FL>77.5**			**Scale age Hatchery FL<77.5**			**Scale age: Hatchery FL>77.5**		
Run Year	O1	O2	O3	O1	O2	O3	O1	O2	O3	O1	O2	O3
85	60	23	0	1	27	2	80	32	0	0	33	15
86	84	81	0	0	42	1	94	60	0	0	33	8
87	224	86	0	3	73	5	95	60	0	0	35	2
88	99	54	0	11	41	0	106	28	0	2	59	1
89	46	49	3	0	46	5	109	51	4	1	79	28
90	21	57	1	0	70	2	62	113	1	4	136	11
91	110	33	0	9	34	4	221	37	0	15	30	2
92	28	27	0	6	17	0	91	32	0	18	41	0
93	31	18	0	2	14	1	48	43	0	5	37	0
94	43	24	0	8	24	1	147	35	0	10	38	2
95	na	na	na	na	na	na	na	na	na	na	na	na
96	na	na	na	na	na	na	na	na	na	na	na	na
97	na	na	na	na	na	na	na	na	na	na	na	na

Table B.3-3: Age composition of wild and hatchery A-run and B-run steelhead at Lower Granite Dam, 1985-1997 run years, based on fork length criteria and scale analysis.

Run Year	**Wild A Run Size at LGR**				**Wild B Run Size at LGR**			
	Total	**O1**	**O2**	**O3**	**Total**	**O1**	**O2**	**O3**
85	17850	11180	6503	167	8858	2494	5986	378
86	17621	7715	9849	57	4369	801	3503	65
87	21847	13689	7969	189	3623	1028	2452	143
88	17429	10215	7214	0	3604	1327	2277	0
89	15928	6430	8860	638	9040	1528	6846	667
90	2922	668	2211	43	6339	607	5591	142
91	15812	10884	4739	188	1510	603	825	82
92	13219	6270	6949	0	6127	2146	3981	0
93	6532	3662	2817	54	821	249	541	31
94	4732	2772	1940	20	2783	1075	1654	54
95	7648	3991	3580	77	342	96	235	10
96	6198	3234	2901	62	1106	312	760	34
97	8474	4422	3967	85	200	56	137	6

Run Year	Hatchery A Run Size at LGR				Hatchery B Run Size at LGR			
	Total	O1	O2	O3	Total	O1	O2	O3
85	na	na	na	na	na	na	na	na
86	72096	43567	28389	141	35897	3725	26359	5814
87	32133	19497	12618	17	13677	1425	11638	614
88	44183	34615	9560	7	21921	3535	18093	293
89	66553	43797	20976	1780	39899	4815	26333	8751
90	25574	8926	16486	162	22030	1804	18873	1353
91	69849	59456	10363	30	11881	4877	6584	420
92	83353	61305	22048	0	25566	9689	15877	0
93	35511	18586	16925	0	16904	3186	13718	0
94	32412	25982	6417	13	7375	2237	4893	245
95	62769	41241	21272	255	8368	1723	6116	529
96	62620	41144	21222	254	13340	2747	9749	844
97	65676	43151	22258	267	12096	2491	8840	765

TAC estimates of LGR run size, Zone 1-6 harvest rates and total run size were compiled for wild and hatchery A-run and B-run steelhead, run years 1985-1997 (Table B.3-4). Wild A-run returns consistently exceeded those of wild B-run, which returned extremely small numbers in 1995 (342 adults to LGR) and in 1997 (200 adults). Harvest rates ranged from 7% to 25% for wild A-run and 24% to 56% for wild B-run. Hatchery A-run returns were consistently greater, and harvest rates lower, than those of hatchery B-run steelhead.

Table B.3-4: Adult run size to Lower Granite Dam (LGR), Zone 1-6 harvest rate, and total run size (LGR run plus harvest) for wild and hatchery A-run and B-run Snake River steelhead (TAC 1997), run years 1985-1997. Harvest rates are based on length method, 1996 and 1997 harvest rate estimates are preliminary.

Run Year	Wild A-Run			Wild B-Run		
	LGR Run Size	Harvest Rate	Total Run Size	LGR Run Size	Harvest Rate	Total Run Size
1985	17850	25.3%	23911	8858	41.1%	15035
1986	17621	14.9%	20696	4369	33.0%	6521
1987	21847	19.4%	27101	3623	47.2%	6863
1988	17429	22.9%	22600	3604	36.6%	5682
1989	15928	17.5%	19296	9040	45.6%	16619
1990	2922	17.2%	3528	6339	28.1%	8812
1991	15812	15.8%	18777	1510	39.1%	2480
1992	13219	16.8%	15881	6127	32.8%	9121
1993	6532	17.2%	7891	821	25.3%	1100
1994	4732	11.0%	5317	2783	24.8%	3701
1995	7648	11.7%	8658	342	28.2%	477
1996	6198	7.2%	6679	1106	56.2%	2525
1997	8474	12.2%	9651	200	23.9%	263

Run Year	Hatchery A-Run			Hatchery B-Run		
	LGR Run Size	Harvest Rate	Total Run Size	LGR Run Size	Harvest Rate	Total Run Size
1985	na	24.9%	na	na	62.5%	na
1986	72096	20.7%	90920	35897	35.4%	55583
1987	32133	33.0%	47943	13677	53.7%	29562
1988	44183	34.9%	67840	21921	49.5%	43432
1989	66553	26.7%	90837	39899	42.2%	69043
1990	25574	25.3%	34256	22030	39.6%	36452
1991	69849	22.4%	89965	11881	41.6%	20358
1992	83353	25.3%	111644	25566	39.3%	42124
1993	35511	30.6%	51198	16904	46.0%	31330
1994	32412	23.3%	42265	7375	27.5%	10179
1995	62769	21.0%	79405	8368	29.0%	11787
1996	62620	16.2%	74726	13340	42.6%	23240
1997	65676	16.8%	78938	12096	11.0%	13591

Numbers of recruits to LGR and total recruits by smolt migration year, 1985-1994, were estimated for wild and hatchery A-run and B-run steelhead (Table B.3-5). Total recruits ranged from 6,400 to 27,100 for wild A-run, and 1,800 to 14,900 for wild B- run, with both groups exhibiting a decreasing trend in numbers over the period (Figure B.3-1). Total recruits ranged from 25,300 to 106,500 for hatchery A-run, and 13,200 to 54,800 for hatchery B-run steelhead.

Table B.3-5: Lower Granite Dam recruits (LGR) and total adult recruits (LGR escapement + harvest) by smolt year (1985-1994) for wild and hatchery A-run and B-run Snake River steelhead.

Smolt Year	Wild A-Run		Wild B-Run		Hatchery A-Run		Hatchery B-Run	
	LGR Recruits	Total Recruits	LGR Recruits	Total Recruits	LGR Recruits	Total Recruits	LGR Recruits	Total Recruits
1985	15684	18946	3252	5839	56192	73779	15657	31504
1986	21541	27109	3972	6763	30837	46198	28269	54070
1987	19118	24032	8315	14874	55754	81997	31221	54811
1988	8829	10683	7201	10715	60313	81899	24107	40279
1989	5407	6435	1431	2198	19288	25303	8388	14266
1990	17887	21338	4615	6959	81504	106110	20754	34517
1991	9107	10958	2741	3991	78243	106532	23652	41727
1992	5678	6690	1913	2548	25258	35487	8609	13404
1993	6415	7235	1344	1834	47509	61095	9197	13172
1994	6978	7742	863	1884	62730	77817	12238	20272

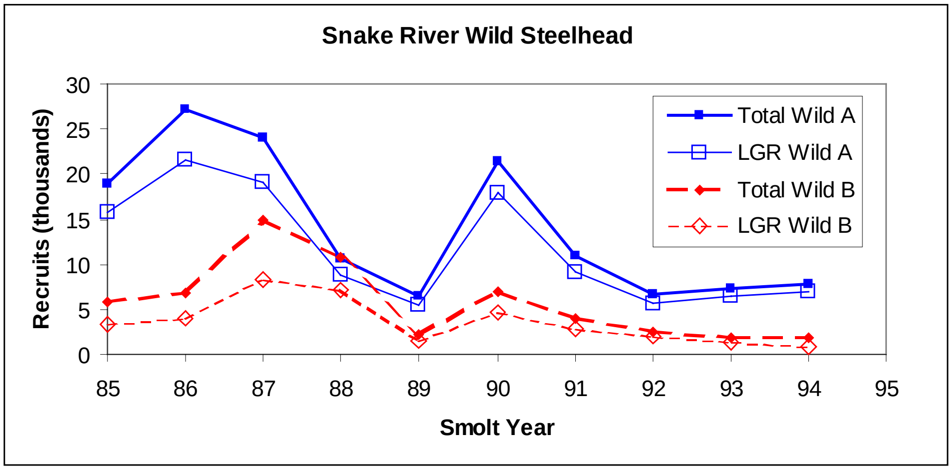

Figure B.3-1: Total recruits and recruits to Lower Granite Dam (LGR) for Snake River wild A-run and wild B-run steelhead, smolt years 1985-1994.

Numbers of wild and hatchery steelhead recruits and SARs (total and upper dam) were estimated for 1985-1994 smolt migrations to update Raymond's (1988) estimates for smolt migrations 1964-1984 (Table B.3-6). Numbers of wild recruits to the upper dam and total wild recruits decreased markedly after the 1969 smolt outmigration, with a temporary increase in the mid-1980s (Table B.3-6, Figure B.3-2). Total wild recruits ranged from 62,000 to 107,000 in the 1960s, and from 5,000 to 46,000 during the 1977-1994 period. Total hatchery recruits increased over time as hatchery smolt releases increased (Table B.3-6).

Table B.3-6: Estimates of steelhead smolt numbers, adult recruits (with and without river harvest), and smolt-to-adult return rates (SAR), 1964-1994 smolt years.

Year of smolt migration	Smolts passing first dam (millions)			Total adult recruits (thousands)			SAR2 (%) (Escapement + Harvest)			Upper dam recruits (thousands)			SAR1 (%) (Escapement)		
	Hatchery	Wild	Combined	Hatchery	Wild	Combined	Hatchery	Wild	Combined	Hatchery	Wild	Combined	Hatchery	Wild	Combined
1964		1.60	1.60		100	100		6.3	6.3		67	67		4.2	4.2
1965		1.50	1.50		85	85		5.7	5.7		55	55		3.7	3.7
1966		1.60	1.60		102	102		6.4	6.4		63	63		3.9	3.9
1967	0.80	1.80	2.60	6	107	113	0.7	5.9	4.3	4	73	77	0.5	4.0	2.9
1968	1.00	1.80	2.80	3	81	84	0.3	4.5	3.0	2	61	63	0.2	3.4	2.3
1969	0.80	1.30	2.10	6	62	68	0.9	4.8	3.2	5	47	52	0.7	3.7	2.4
1970	2.60	1.60	4.20	38	54	92	1.5	3.4	2.2	29	41	69	1.1	2.6	1.7
1971	3.20	1.80	5.00	26	56	82	0.8	3.1	1.6	19	41	60	0.6	2.3	1.2
1972	1.40	1.10	2.50	7	21	28	0.5	1.9	1.1	6	17	22	0.4	1.5	0.9
1973	2.50	1.30	3.80	3	9	12	0.1	0.7	0.3	3	8	11	0.1	0.6	0.3
1974	3.60	1.40	5.00	5	18	23	0.1	1.4	0.5	5	17	21	0.1	1.3	0.5
1975	2.40	0.80	3.20	45	17	62	1.9	2.1	1.9	39	15	54	1.7	1.8	1.7
1976	1.80	1.40	3.20	28	28	56	1.6	2.0	1.8	24	24	48	1.4	1.7	1.5
1977	0.90	0.50	1.40	7	5	12	0.8	1.0	0.9	6	5	11	0.7	0.9	0.8
1978	1.20	0.90	2.10	17	30	47	1.4	3.3	2.2	16	28	44	1.3	3.1	2.0
1979	1.50	1.10	2.60	31	37	68	2.1	3.4	2.6	29	35	64	2.0	3.2	2.4
1980	2.60	1.00	3.60	55	27	82	2.1	2.7	2.3	52	25	77	2.0	2.5	2.2
1981	2.40	1.30	3.70	29	15	44	1.2	1.2	1.2	27	14	41	1.1	1.1	1.1
1982	3.30	1.00	4.30	167	40	207	5.1	4.0	4.8	141	34	175	4.3	3.4	4.0
1983	2.10	0.80	2.90	67	27	94	3.2	3.4	3.2	52	21	73	2.5	2.6	2.5
1984	3.20	1.00	4.20	151	46	197	4.7	4.6	4.7	120	37	157	3.7	3.7	3.7
1985	3.08	0.63	3.71	105	25	130	3.4	4.0	3.5	72	19	91	2.3	3.0	2.4
1986	3.87	0.82	4.70	100	34	134	2.6	4.1	2.9	59	26	85	1.5	3.1	1.8
1987	3.33	0.74	4.07	137	39	176	4.1	5.2	4.3	87	27	114	2.6	3.7	2.8
1988	5.61	0.80	6.41	122	21	144	2.2	2.7	2.2	84	16	100	1.5	2.0	1.6

Year of smolt migration	Smolts passing first dam (millions)			Total adult recruits (thousands)			SAR2 (%) (Escapement + Harvest)			Upper dam recruits (thousands)			SAR1 (%) (Escapement)		
	Hatchery	Wild	Combined	Hatchery	Wild	Combined	Hatchery	Wild	Combined	Hatchery	Wild	Combined	Hatchery	Wild	Combined
1989	6.35	0.74	7.09	40	9	48	0.6	1.2	0.7	28	7	35	0.4	0.9	0.5
1990	7.35	0.94	8.30	141	28	169	1.9	3.0	2.0	102	23	125	1.4	2.4	1.5
1991	7.02	0.78	7.79	148	15	163	2.1	1.9	2.1	102	12	114	1.5	1.5	1.5
1992	4.73	0.72	5.45	49	9	58	1.0	1.3	1.1	34	8	41	0.7	1.1	0.8
1993	8.03	0.72	8.75	74	9	83	0.9	1.3	1.0	57	8	64	0.7	1.1	0.7
1994	5.81	0.64	6.45	98	10	108	1.7	1.5	1.7	75	8	83	1.3	1.2	1.3
1995	7.74	0.60	8.34												
1996	7.88	0.65	8.53												

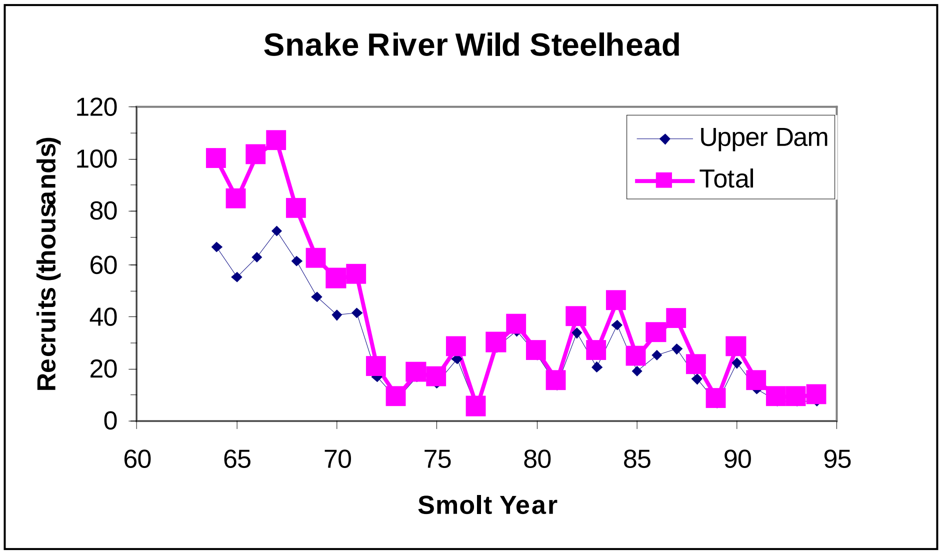

Figure B.3-2: Total recruits and recruit to upper dam for Snake River aggregate wild steelhead, smolt years 1964-1994.

Smolt-to-adult return rates of Snake River wild steelhead decreased after the 1960s (Table B.3-6, Figure B.3-3). SAR2 estimates ranged from 4.5% to 6.3% in the 1960s, and from 1.0% to 5.2% during the 1977-1994 period (Table B.3-6, Figure B.3-3). SAR1 estimates ranged from 3.4% to 4.2% in the 1960s, and from 0.9% to 3.7% during the 1977-1994 period (Table B.3-6, Figure B.3-3). Because of harvest rate differences between stock groups, recent values of SAR1 for wild B-run steelhead were likely less than for the aggregate. Hatchery steelhead SARs have been generally similar to those of wild steelhead since the mid-1970s (Table B.3-6).

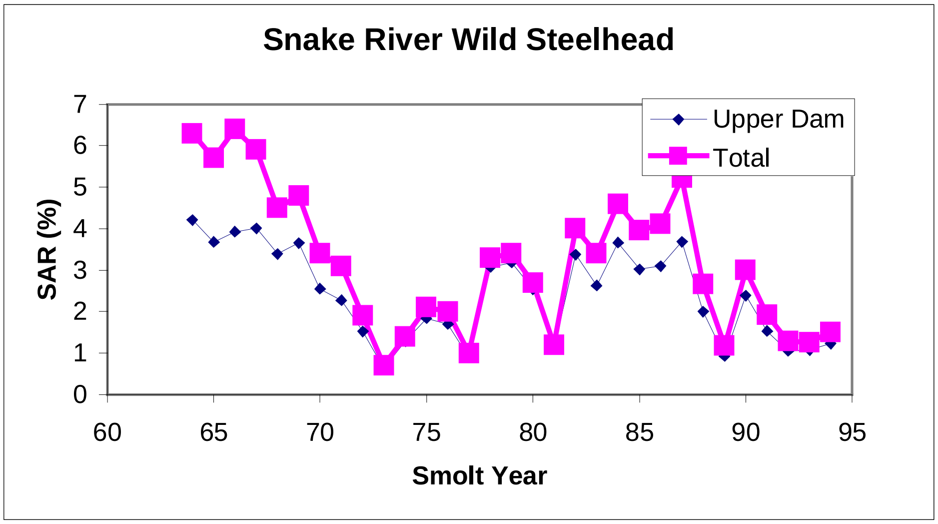

Figure B.3-3: Smolt-to-adult return rates for total recruits (SAR2) and recruits to upper dam (SAR1) for Snake River aggregate wild steelhead, smolt years 1964-1994.

Smolt to-adult return rates decreased substantially for both Snake River wild steelhead and wild spring/summer chinook since 1969 with construction and operation of the FCRPS, but spring/summer chinook suffered the greater decreases (Table B.3-7). For recruits to the upper dam (SAR1), the ratio of historic SAR to recent SAR was 4.13 for steelhead compared to 6.91 for spring/summer chinook. For total recruits (SAR2), the ratio of historic SAR to recent SAR was 4.18 for steelhead compared to 11.21 for spring/summer chinook.

Table B.3-7: Snake River steelhead and spring/summer chinook SAR comparisons, historic and recent smolt migration years, and ratio of steelhead SAR to chinook SAR, 1964-1994. Historic period is defined as pre-1970 smolt years; recent period is 1992-1994 smolt years for both species. ND--no data.

	Historical Increment Comparison				**Ratio Comparison**
Smolt	**SAR1 to Upper Dam**		**SAR2 Escapement Catch**		**Annual Ratio**
Year	***Steelhead***	***Chinook***	***Steelhead***	***Chinook***	***(SH/CH)***
1964	4.21%	2.35%	6.30%	4.40%	1.43
1965	3.68%	2.32%	5.70%	4.13%	1.38
1966	3.93%	2.31%	6.40%	3.73%	1.72
1967	4.01%	4.49%	5.90%	7.25%	0.81
1968	3.39%	2.58%	4.50%	4.24%	1.06
1969	3.66%	3.83%	4.80%	6.59%	0.73
1970	2.55%	1.92%	3.40%	3.85%	0.88
1971	2.27%	1.53%	3.10%	2.80%	1.11
1972	1.52%	1.02%	1.90%	1.21%	1.57
1973	0.63%	0.49%	0.70%	0.50%	1.39
1974	1.29%	1.39%	1.40%	1.53%	0.92
1975	1.84%	3.11%	2.10%	3.64%	0.58
1976	1.70%	0.92%	2.00%	0.98%	2.05

Smolt	Historical Increment Comparison				Ratio Comparison
	SAR1 to Upper Dam		SAR2 Escapement Catch		Annual Ratio
1977	0.90%	0.35%	1.00%	0.37%	2.74
1978	3.07%	0.98%	3.30%	1.03%	3.22
1979	3.18%	1.09%	3.40%	1.18%	2.89
1980	2.54%	0.55%	2.70%	0.59%	4.61
1981	1.11%	1.39%	1.20%	1.50%	0.80
1982	3.37%	1.70%	4.00%	1.83%	2.18
1983	2.63%	1.83%	3.40%	1.97%	1.73
1984	3.66%	2.56%	4.60%	2.76%	1.66
1985	3.02%	ND	3.95%	ND	ND
1986	3.10%	ND	4.11%	ND	ND
1987	3.68%	ND	5.23%	ND	ND
1988	2.00%	ND	2.67%	ND	ND
1989	0.93%	ND	1.17%	ND	ND
1990	2.38%	ND	3.00%	ND	ND
1991	1.53%	ND	1.93%	ND	ND
1992	1.05%	0.19%	1.28%	0.20%	6.35
1993	1.08%	0.38%	1.26%	0.39%	3.20
1994	1.23%	1.02%	1.51%	1.05%	1.44
Geomean:	***Steelhead***	***Chinook***	***Steelhead***	***Chinook***	***(SH/CH)***
Historic SAR	**3.80%**	**2.87%**	**5.55%**	**4.89%**	**1.13**
Recent SAR	**1.12%**	**0.42%**	**1.34%**	**0.44%**	**3.08**
Incremental Change	3.41	6.91	4.18	11.21	

B-4 Discussion

The similarity in historic survival rate patterns between Snake River steelhead and spring/summer chinook since FCRPS development and operation lends support to NMFS (1998) conclusion that hydropower management options would likely have similar effects on both species. Smolt-to-adult return rates (SAR2) were significantly correlated between wild steelhead and wild spring/summer chinook for the 1964-1994 smolt migrations (Figure B.4-1), suggesting that both species were influenced in the past by common human-caused and environmental factors.

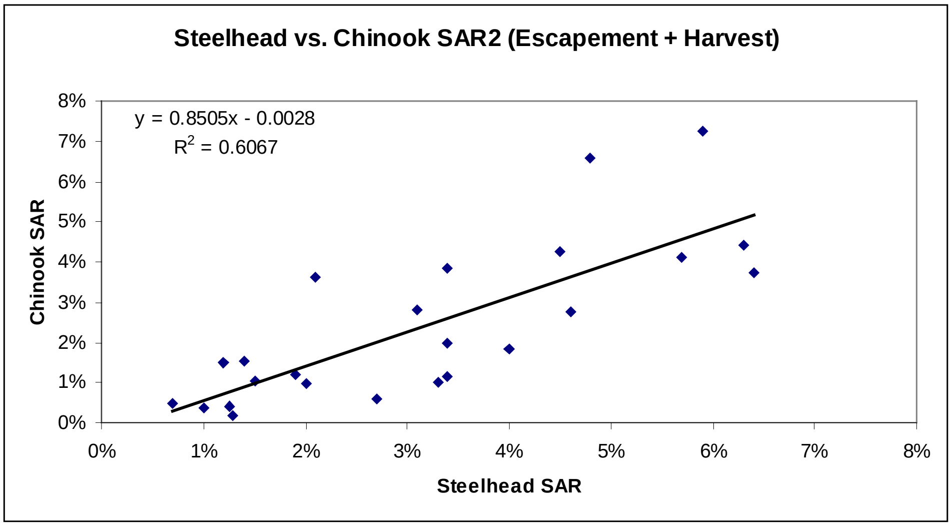

Figure B.4-1: Regression of SAR2 (total recruits) estimates for Snake River wild steelhead and wild spring/summer chinook, 1964-1994 smolt years.

Smolt-to-adult return rate (including harvest; SAR2) for each species was significantly correlated with the water travel time (WTT) for migration years 1964-1994 (Figure B.4-2). Predicted SAR2 was higher for steelhead than for spring/summer chinook across the observed range of water travel times. Water travel time (Lewiston to Bonneville Dam) is a function of the number and volume of reservoirs, and flow. WTT during 1964-1969 with 4-6 dams ranged from 5 to 17 days, depending on the number of reservoirs and flow. In contrast, WTT with 8 dams (1975-1994) ranged from 13 to 39 days, depending on flow.

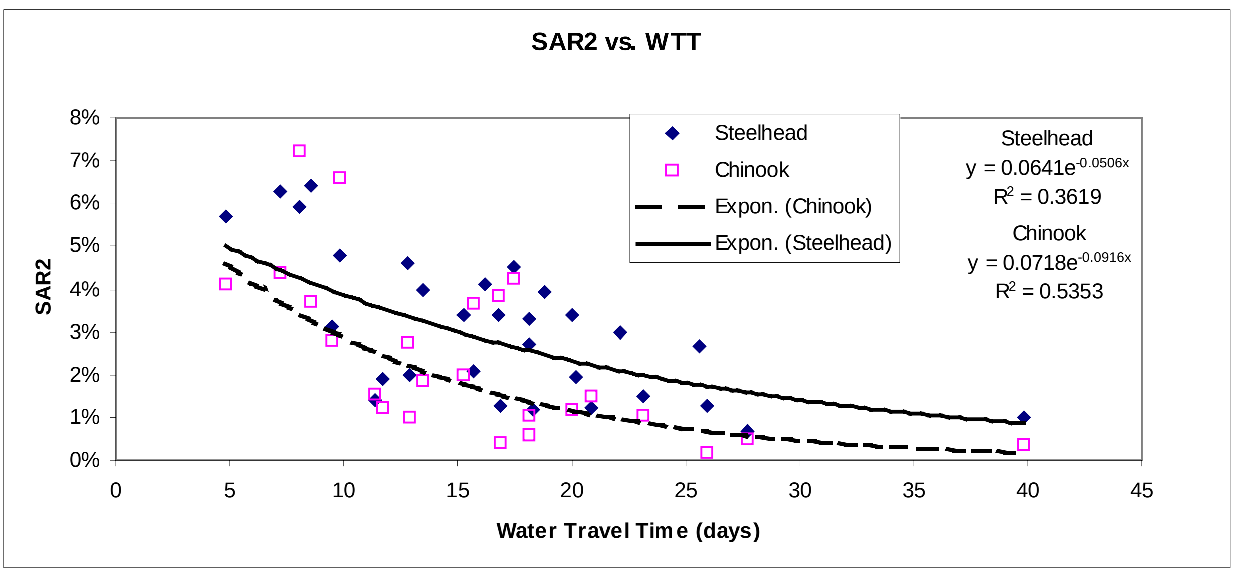

Figure B.4-2: Regressions of SAR2 (total recruits) estimates for wild Snake River steelhead and spring/summer chinook with ater travel times, 1964-1994 smolt migrations.

There are several mechanisms by which water travel time may affect smolt survival rates (summarized under the hydro hypothesis in Marmorek and Peters 1998). Water travel time directly influences fish travel time. Low water velocity may result in migration delay, poorly synchronized estuary entry, reduced fish condition, greater metabolic cost, and greater rate of residualism (for steelhead). In addition, low water velocity contributes to cumulative effects in the reservoirs (longer exposure to higher temperatures and, consequently, increased vulnerability to stressors and predation) and at dams (i.e., low flow years have had less spill). In recent years, a majority of smolts have been transported, and may avoid some of these influences of slack water. However, Mundy et al. (1994) concluded "[j]uvenile salmon die at rates related to physical conditions in the river, including the hydroelectric system, despite the transportation effort." The association of SAR and WTT for both spring/summer chinook and steelhead provides additional evidence for this conclusion and the hydro hypothesis.

A limitation of the empirical SAR vs. WTT relationship is the confounding of water velocity with other factors over time, such as number of dams, water management, turbine installation and system improvements. Summarizing the data by period (Figure B.4-3) reduces this confounding. The recent (1985-1994) pattern of SAR vs. WTT appears consistent with the pattern for the 1977-1984 migrations, and the overall pattern (Figure 4.B-3).

Trends in historic WTT are a function of dam construction (Figure B.4-4). Since the last dam in 1975, the highest flow years (1976, 1984, 1982) resulted in slower WTT (and lower SAR) than before John Day Dam construction in 1968. To restore a WTT range that was associated with higher SARs, will require restoring free-flowing river reaches (Options A3, A5, B1, B2, C1 and C2).

Estimates of smolt numbers and SAR for 1985-1994 are moderately sensitive to assumed values of FGE and spill efficiency at Lower Granite Dam. Krasnow (1998) estimated FGE to be 0.74 for 1985-1990 and 0.81 after 1990, compared to Petrosky (1998) preliminary SARs which were based on 0.70 FGE assumption for all years. The Krasnow FGE estimates changed the preliminary SARs by factors of 1.06 for 1985-1990 and 1.16 after 1990. These changes in estimated SAR were not large enough to alter conclusions about historical patterns of SAR for steelhead, or relative change in SAR between steelhead and spring/summer chinook.

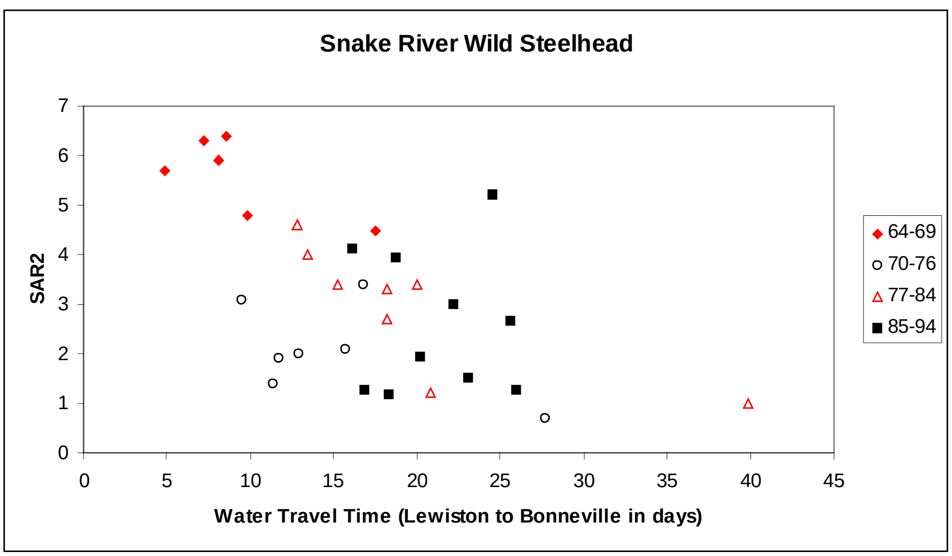

Figure B.4-3: Smolt-to-adult return rate (escapement + harvest) of Snake River wild steelhead, 1964-1994 smolt migrations by time periods.

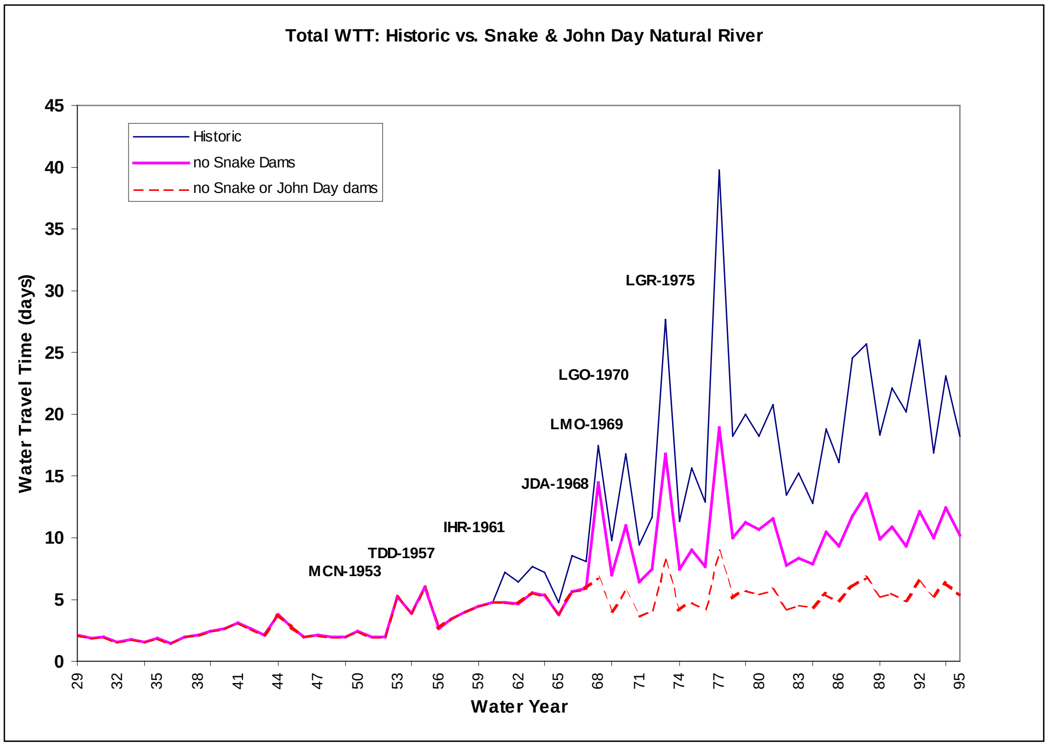

Figure B.4-4: Historic water travel time (WTT), and WTT that would have occurred without the Lower Snake dams, and without Snake and John Day dams, 1929-1995 (historic flows assumed). Dates of dam completion are inset.

Separate SAR estimates for A-run and B-run steelhead would help alleviate an inherent weakness of the aggregate stock approach. That is, populations within an aggregate with different productivity and management effects may not respond in the same way as the aggregate. Survival and recovery standards for Snake River spring/summer chinook account for this by explicitly requiring a “high percentage” (interpreted as 80%) of index stocks achieve the standards. Among Snake River steelhead, the B-run group clearly represents the weaker stock group (Table B.3-4) and substantially more than 20% of the total production potential. Inability to partition smolt estimates into A-run and B-run components currently prevents estimating A-run and B-run SARs, without additional assumptions. However, the relationship between SAR1 and SAR2 for the two groups is a straightforward calculation based on harvest rate estimates. If SAR2 (escapement + catch) is assumed equal for A-run and B-run, then SAR1 (escapement) for B-run will be less than the aggregate SAR1 (and A-run SAR1) because of the higher harvest rates. For example, if SAR2 were 2% for each, and harvest rates were 0.15 for A-run and 0.35 for B-run, the respective SAR1 estimates would be 1.7% and 1.3%. If SAR2 were less for B-run stocks, as might be hypothesized by their longer ocean residence, a larger difference in SAR1 between the groups would be expected. The feasibility of partitioning SAR for the two wild stock groups should be explored in FY99.

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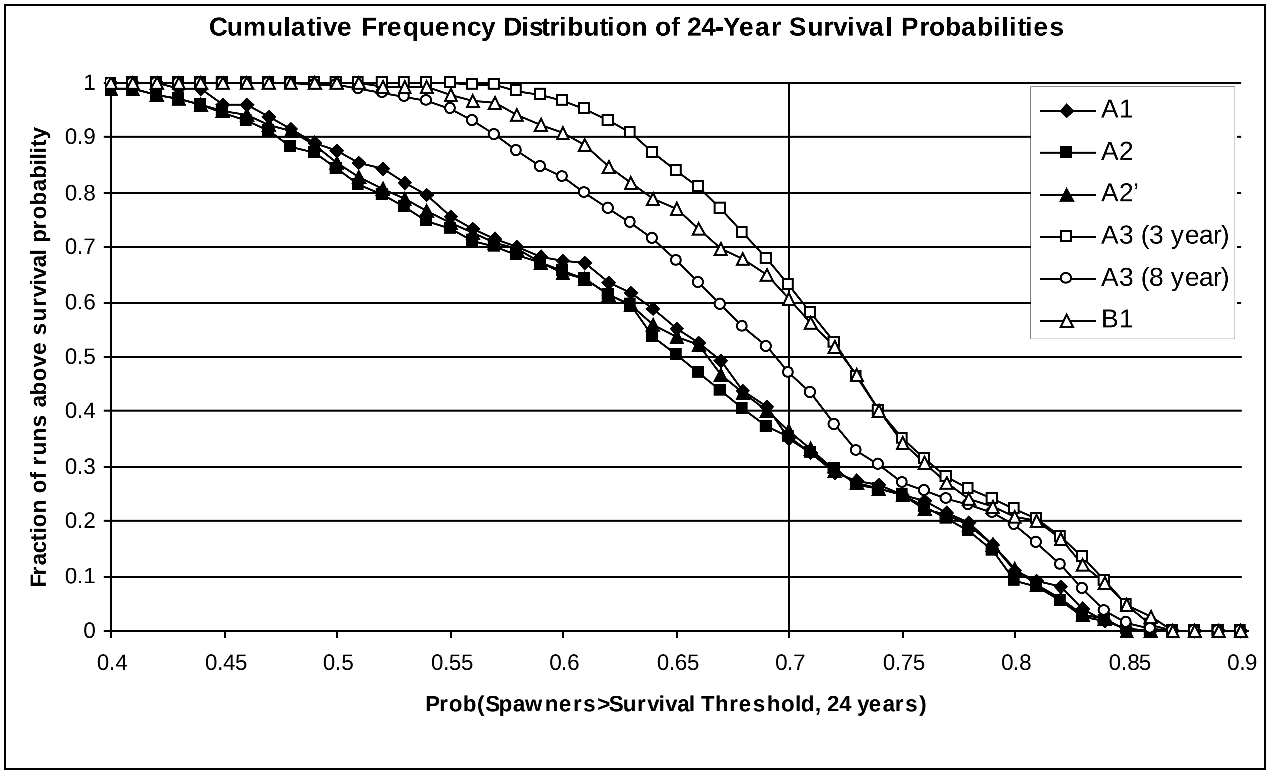
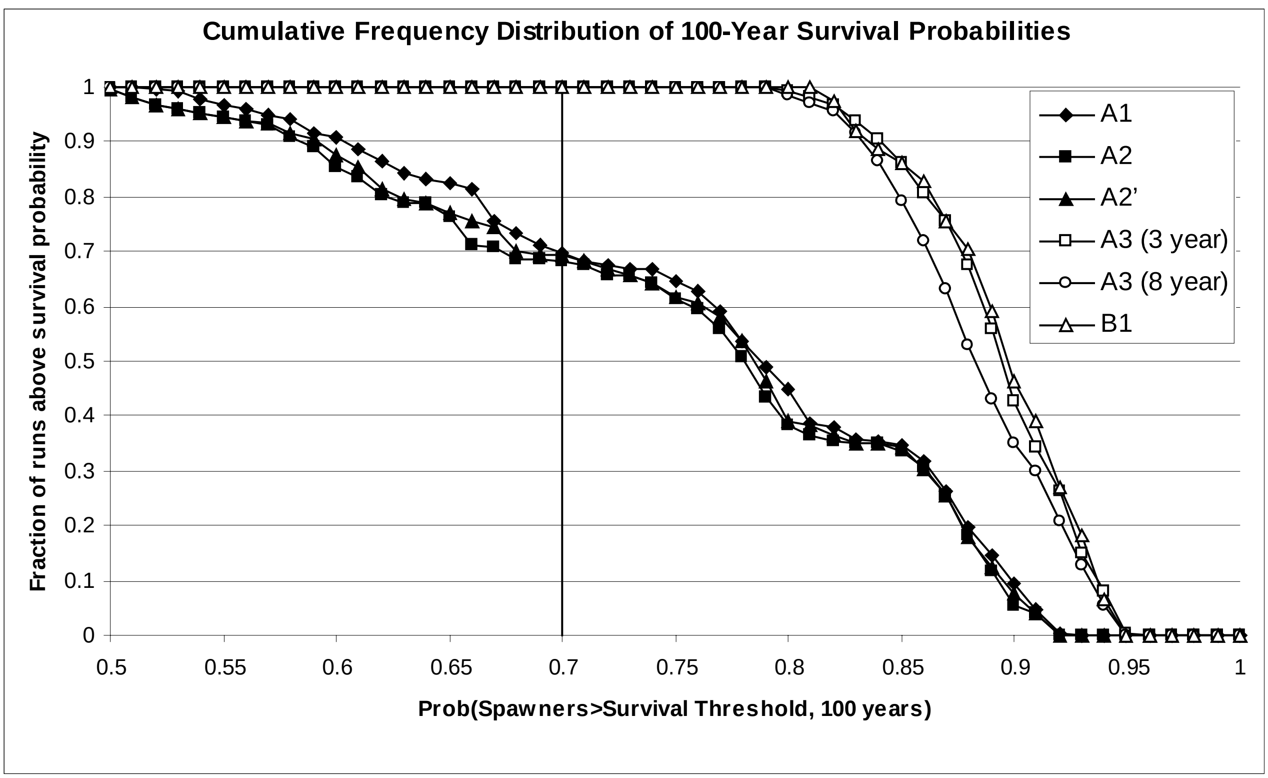
Appendix C: Cumulative Frequency Distributions for Spring/Summer and Fall Chinook

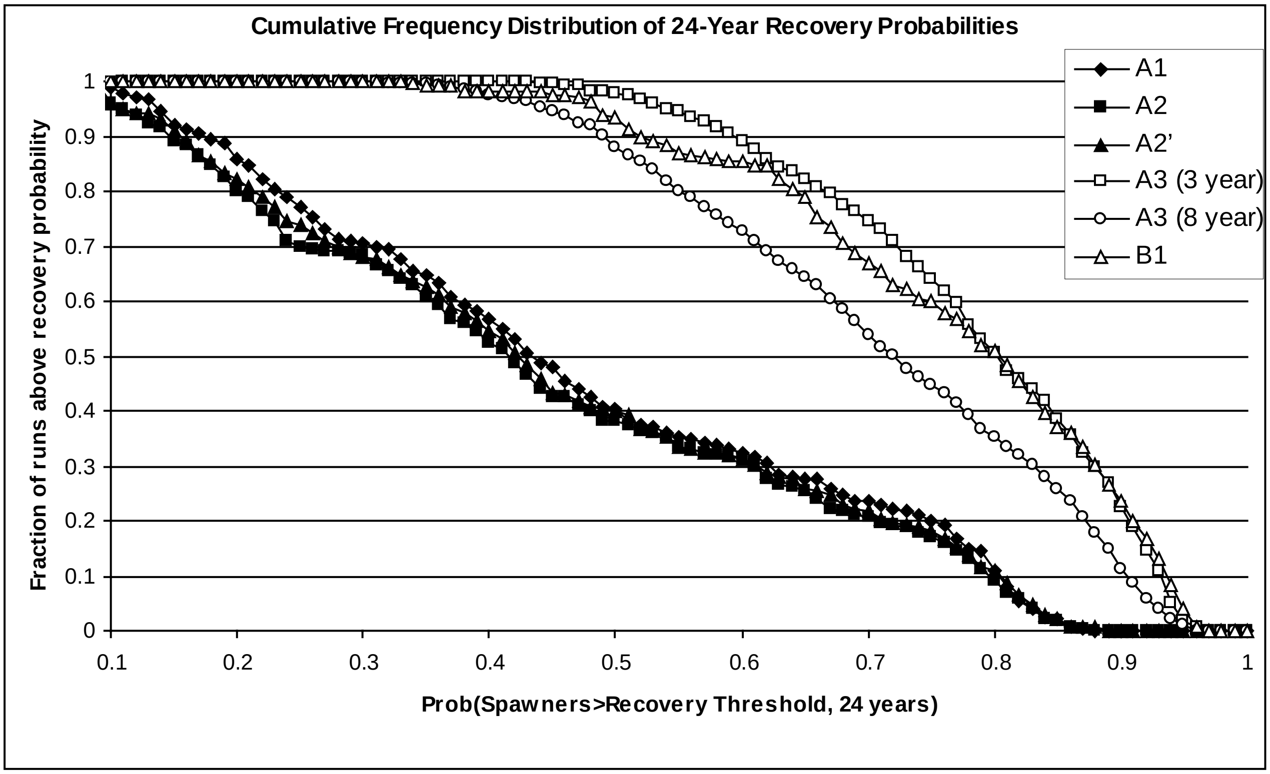
This Appendix shows cumulative frequency distributions for all actions and standards. The distributions assume equal weights on all combinations of hypotheses. Results for spring/summer chinook are shown in Figure C-1; preliminary results for fall chinook are shown in Figure C-2. We use this method of displaying results because it provides a broad picture of the potential risks and benefits to the stocks of a given action. The cumulative frequency distribution can be explained as follows. The range of survival probabilities that resulted from each action are displayed along the bottom axis. For each of these values, the point on the graph shows the fraction of all of the runs for that action in which that particular survival probability was equaled or exceeded. For example, 0.84 of all of the spring/summer chinook runs for A1 produced a 24-year survival probability that was 0.52 or higher (top panel, Figure C-1). A more detailed explanation of these graphs is provided in section 2.2.2.

The vertical lines show the NMFS standards (0.7 for the survival standards, 0.5 for the 48-year recovery standard). However, the graphs also show the fraction of the runs that meet any standard. For example, one may wish to apply a more risk-adverse standard of 0.8 for the 24-year survival standard for spring/summer chinook (top panel of Figure C-1). In this case, the fraction of A1, A2, and A2' runs meeting this standard is approximately 0.1, and for A3 and B1 is approximately 0.2. Conversely, one could apply a less risk-adverse standard of 0.6. In this case, around 0.65 of the runs for A1, A2, and A2' achieve this alternative standard, while 0.97 of the A3 (3-year) runs, 0.8 of the A3 (8-year) runs, and 0.9 of the B1 runs meet this standard. These graphs therefore provide a way to explore the performance of the actions relative to different levels of risk.

Spring/Summer Chinook







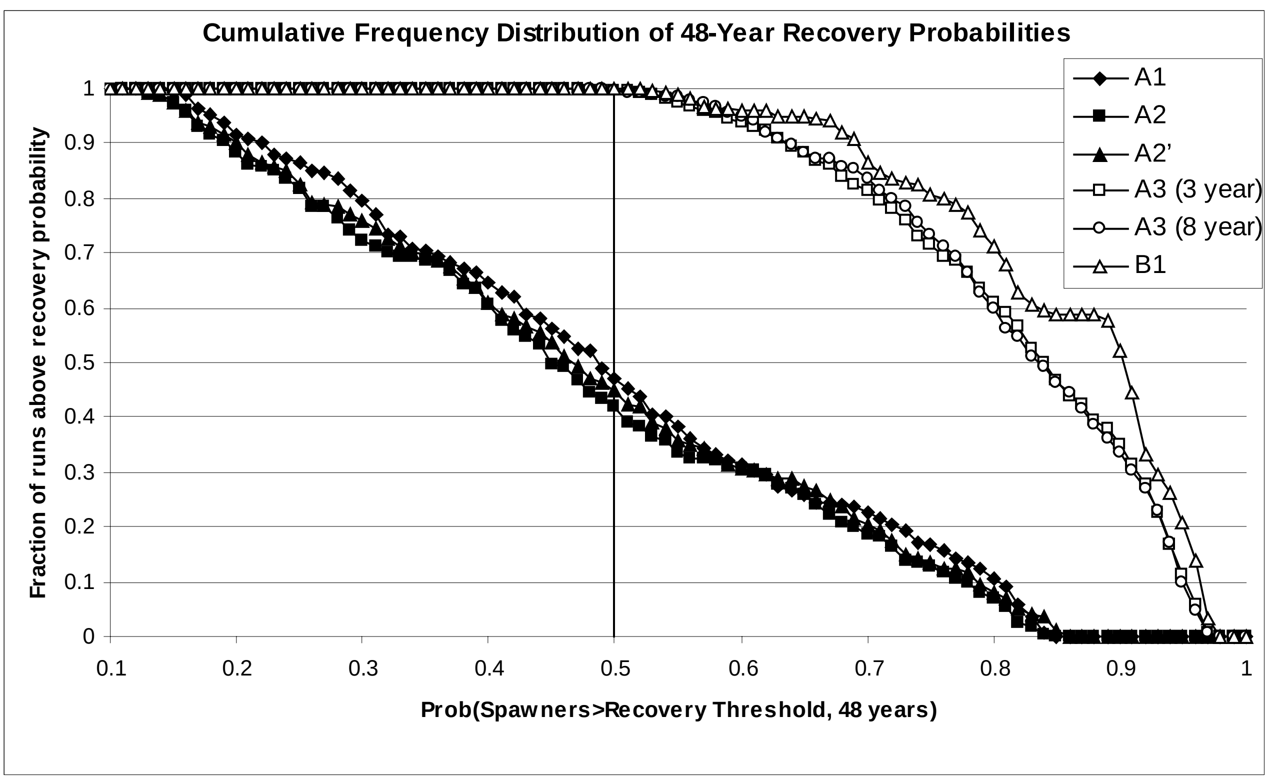
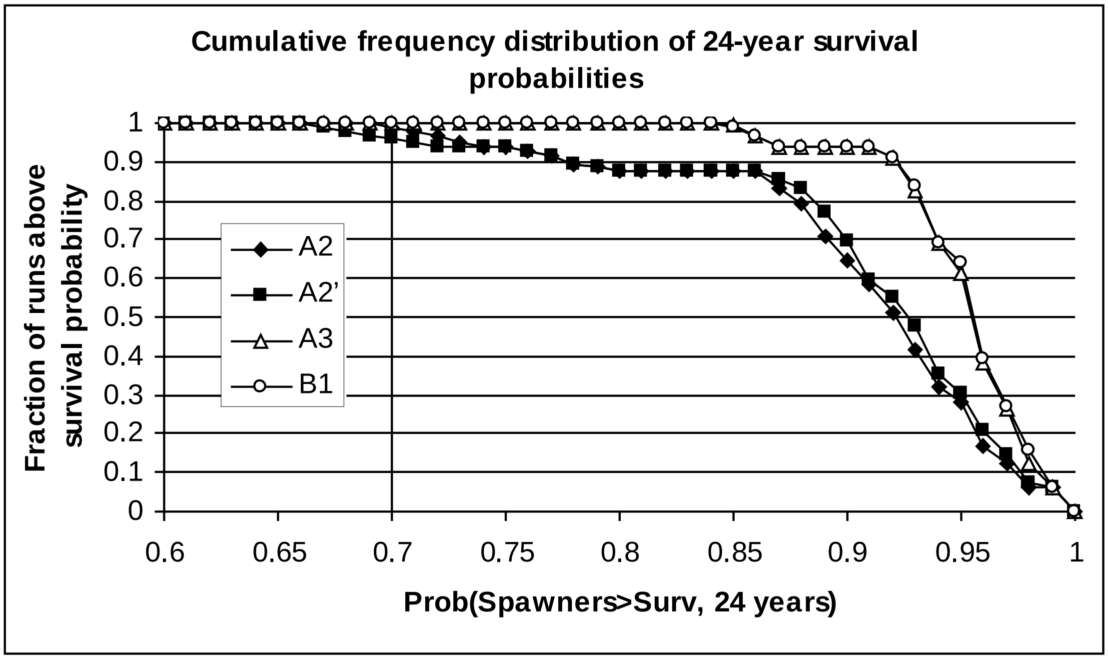
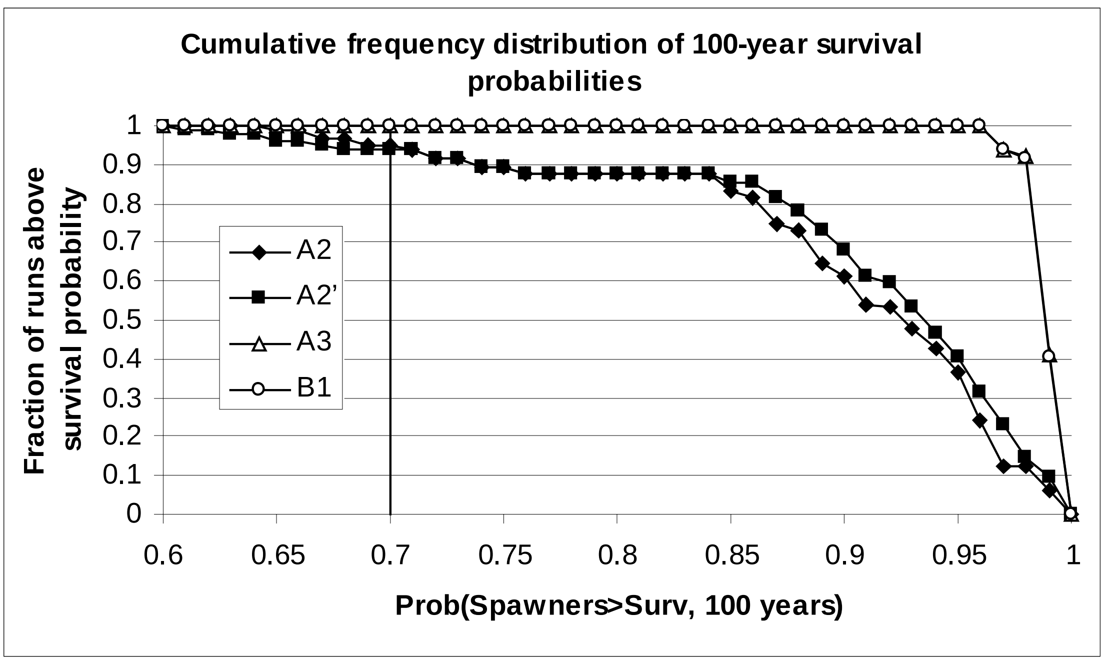
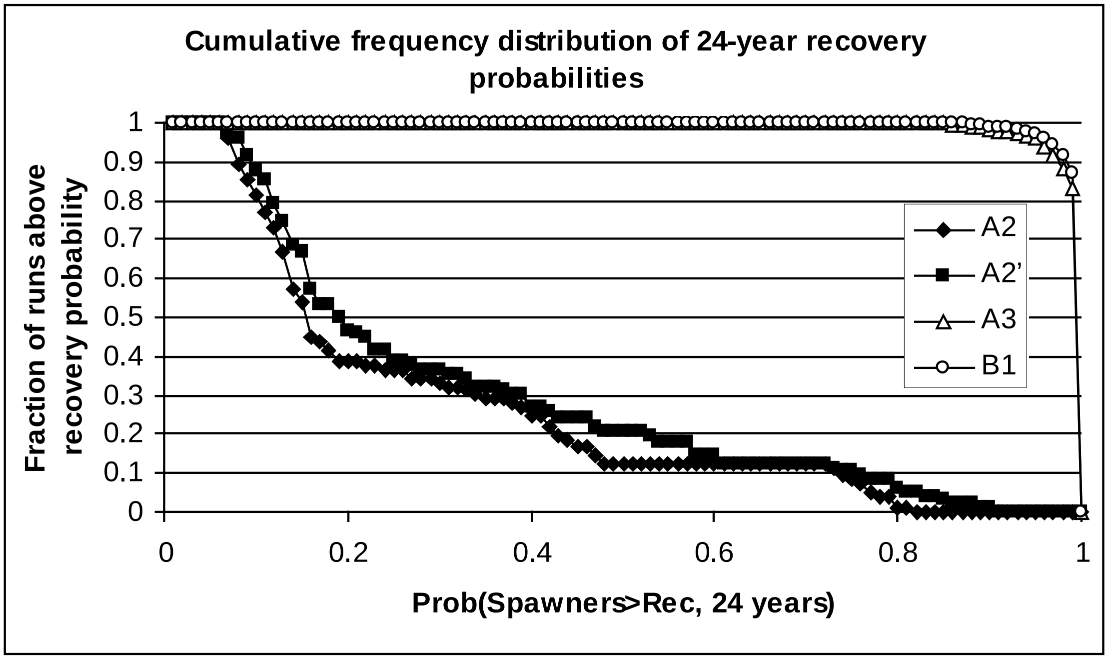


Figure C-1: Cumulative frequency distributions of survival and recovery probabilities for sping/summer chinook. Critical levels used by NMFS to define the Jeopardy Standards are indicated by a vertical line.

Fall chinook (preliminary results)







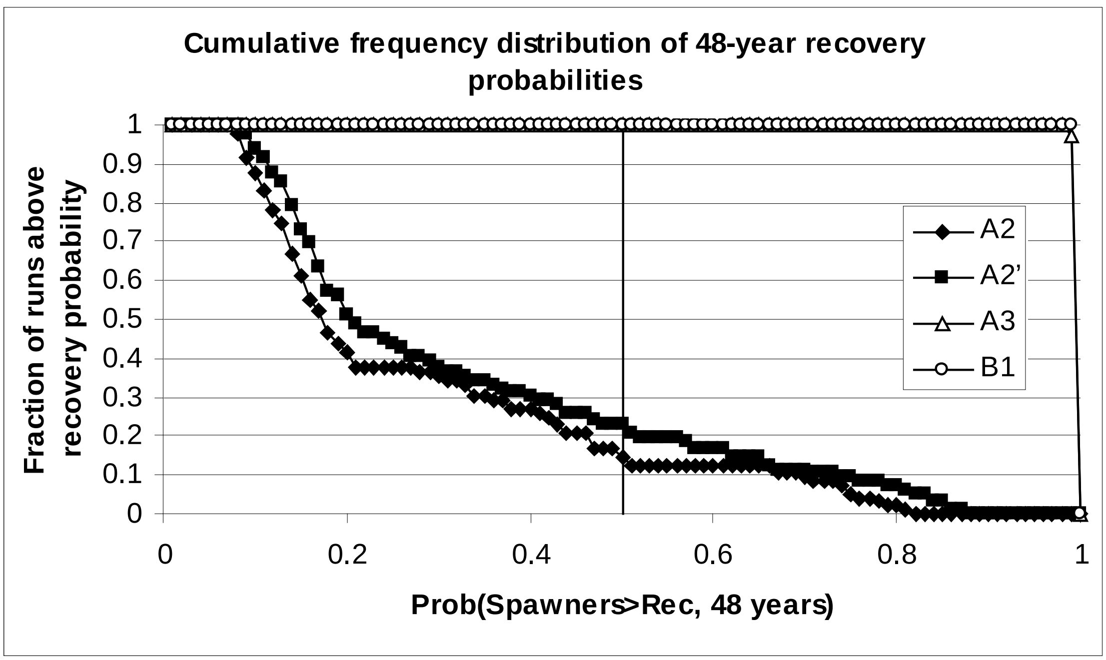


Figure C-2. Cumulative frequency distributions of the 4 jeopardy probabilities for actions A2, A2', A3, and B1 for fall chinook (preliminary results). Critical levels used by NMFS to define the Jeopardy Standards are indicated by a vertical line.