An Assessment of Lower Snake River Hydrosystem Alternatives on Survival and Recovery of Snake River Salmonids

Appendix ___ to the U.S. Army Corps of Engineers' Lower Snake River Juvenile Salmonid Migration Feasibility Study

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1.0 Executive Summary

1.1 Overview of Analytical Approach and Constraints

Sockeye salmon, spring/summer chinook salmon, fall chinook salmon, and steelhead from the Snake River have been listed under provisions of the U.S. Endangered Species Act. This report provides a biological evaluation of management alternatives for the Federal Columbia River Power System (FCRPS) in the context of providing for the survival and recovery of these threatened and endangered species. The report provides a scientific assessment of the likely effects and risks associated with alternative management options, but is not intended to make recommendations about these alternative actions.

The conceptual core of this analysis is a life-cycle model that traces these salmon populations from egg deposition through incubation, freshwater rearing and down-river passage, growth and survival in the ocean, and then the return of spawners upriver to complete the cycle. Threats to survival and, conversely, opportunities for recovery, occur at every stage of this life cycle. In addition, because of the tremendously wide range of habitats and large areas traveled by these species, the problem is one of ecosystem management as opposed to single threat abatement. The primary data that feed into the analysis are time series of fish numbers in different life stages (and observed at different portions the hydrosystem), as well as more focused experimental studies using marked and PIT-tagged fish. There are large gaps in these data, with substantial uncertainties – a situation that precludes arriving at a clear-cut answer through a simple analysis. To meet the challenge of data gaps and contentious scientific uncertainties, the National Marine Fisheries Service (NMFS) Northwest Fisheries Science Center (NWFSC) relied heavily on a multi-agency, multi-participant process called the PATH process. This process was designed in part to allow a wide community of scientists and managers to propose hypotheses for consideration. The mechanics of the analysis embedded in the PATH process are technically difficult, but important to understand in principle.

PATH adopts the above life cycle model as its core analytical tool. It then applies the model to historical data and in so doing:

- Establishes estimates of historical trends in reproduction and components of survival (such as in-river survival during downstream migration);

- Generates hypotheses about sources of mortality that might account for the portion of salmonid declines that cannot be explained by direct estimates of mortality occurring in or caused by the migration corridor; and

- Generates estimates of variability in the underlying processes.

Then, in the second stage of its analysis, PATH uses the same life-cycle model to examine the outcome of different management options by running a large set of future scenario simulations under different management actions. These future simulations are exhaustively examined in light of sensitivity analyses (different visions of how factors outside of the hydrosystem might change in the future). The uncertainty in model output comes from the inherent uncertainty of a variable environment (no one can be sure whether next year will be a drought year or a high rainfall year) and from different assumptions invoked when running the model. Because there are so many possible hypotheses regarding the life-cycle model and future scenarios, PATH runs simulations tailored to all of the plausible hypotheses put forth by participants. The result is a large set of "outcomes", with each outcome corresponding to a range of stochastic (varied) population projections into the future. Although PATH examines as many as 6 or 7 different management options, for clarity, this report focuses primarily on comparisons between the breaching of four dams on the lower Snake River versus no breaching (but transportation of fish in barges). In this Executive Summary, these options are simply referred to as "breaching" versus "transportation".

Given a large set of different combinations of assumptions (ranging from 240 to 1,920) and variable output under each assumption set (depending on chance and different scenarios for future actions), there is an overwhelming richness of information to distill. NMFS is using the relative probability of meeting survival and recovery criteria to summarize the likely effects of alternative management actions. The actual criteria depend on the characteristics of each stock and its natal stream, but can be thought of as requirements that a low population threshold is exceeded in more than 70% of the years (survival criterion), and that an upper population threshold is achieved within 48 years (recovery criterion). For any specific assumption set and management action, PATH simulations produce a fraction of Monte Carlo simulations that satisfy these recovery and survival criteria (this fraction can be thought of as a relative probability of succeeding). By averaging these fractions or relative probabilities over <u>all</u> assumption sets we obtain an average measure of an action's success. An alternative way of summarizing the same data is to ask over what percentage of assumption sets are survival and recovery criteria met. As a <u>hypothetical</u> example, simulations from the PATH process might indicate that with breaching a species met survival criteria under 40% of the assumption sets and with transportation the same species met survival criteria under 15% of the assumption sets. This would yield the hypothetical conclusion that breaching was successful under 25% more of the assumption sets than was transportation. The difference between these percentages for alternative management options could depend on the harvest scenarios that are projected for the future (in a sensitivity analysis) or on exactly what assumption sets were implemented in the models.

The point of these somewhat complicated "measures" is to assess the "robustness" of salmonid performance under different management actions. Such robustness measures point towards "risk averse" management or management that is more likely to "work" under the widest range of hypotheses about how the ecosystem functions. Because of the current levels of uncertainty (due to lack of data and our inability to predict with accuracy future ocean climate conditions) about these salmonid populations, NMFS concludes that focusing on risk averseness is a logical approach. However, there is an urgent need to expend greater effort on identifying which hypotheses are best supported by empirical data. Fortunately, the PATH process provides analytical output that directs attention towards those hypotheses that most influence outcomes and that thus need rigorous empirical scrutiny. As the PATH process evolves, NMFS recommends that PATH more assertively discard hypotheses for which there is only weak empirical support and combine hypotheses that have negligible impact on outcomes; if the assumption set is narrowed, decision analyses will become increasingly transparent. In this report, NMFS makes an effort to provide suggestions to PATH about priorities for analysis. At this stage, however, NMFS concludes that the broad exploration of uncertainty evident in PATH analyses to date has been an appropriate approach. It is worth noting that although there may be uncertainty about particular parameter values used when running PATH simulations, scientific consensus has been achieved on the core life cycle model, and on the techniques used to reconstruct population times series from observations of redds (nests) and spawners.

1.2 A Technical Sketch of Key Uncertainties

The decline of salmonid populations in the Snake River and elsewhere in the Pacific Northwest coincided with a broad range of extensive environmental changes, including the construction of dams, massive degradation of habitat quality, increased withdrawal of water for irrigation, expansion of hatchery releases, and so forth. Although the construction of dams is perhaps the most visible threat to Snake River salmon, it clearly is not the only threat. Thus, a major uncertainty concerns the degree to which aspects of the ecosystem other than hydropower have contributed significantly to declines in salmonid populations. Because it is not possible to go back in history and do experiments, one cannot unambiguously conclude where management should turn for salmon recovery. Instead, inferences linking salmon recovery to alternative management actions require intricate statistical procedures that attempt to disentangle the many covarying factors that coincide with salmonid declines.

A couple of key technical concepts surrounding this uncertainty are worth elaborating.
The first concept concerns fish that are transported in trucks and barges to below Bonneville Dam as a means of circumventing direct mortality due to passage at the hydroprojects. These transported fish generally survive well in the trucks and barges and, if enough fish could be collected for transportation, many of the negative effects of dams might be mitigated. The key question is how well transported fish survive below Bonneville Dam, after they are released and continue their life cycle in the ocean. To quantify the possibility that transported fish suffer a delayed effect due to truck or barge transportation, scientists focus on ***differential delayed transportation mortality*** (or what is measured by the "D-value" in technical jargon); differential delayed transportation mortality refers to the potentially lowered survival of transported fish after they are released below Bonneville Dam compared to fish that were not transported. Early estimates of this differential delayed transportation mortality were high, but estimates for recent years using improved methods provided by PIT-tag technology indicate that spring/summer chinook salmon do not suffer major consequences from being transported in terms of a differential delayed mortality. Ongoing experiments by NMFS are likely to resolve the uncertainty regarding differential delayed transportation mortality in 5 to 10 years.

A second uncertainty concerns the possible effect of climatic conditions in the estuary and ocean on all salmonids (regardless of where the populations reside relative to hydrosystem projects). If ocean conditions remain poor or deteriorate considerably, then the opportunities for recovery will be limited, whereas if ocean conditions improve broadly, then populations may recover without any major management actions. This uncertainty might be viewed as the extent to which one views the future optimistically or pessimistically, and it is an uncertainty that cannot be easily resolved by science.

A final major uncertainty associated with all discussion of hydrosystem actions and salmonid recovery involves what scientists have labeled "***extra mortality***". Historically, a much larger percentage of the fish that left the Snake River as juveniles later returned to spawn as adults than is currently the case. Even after scientists account for direct losses due to observable mortality at dams or in reservoirs and declines due to ocean conditions that impact all salmonid populations, there seems to be a large "unexplained" mortality occurring below Bonneville Dam that must be invoked to account for the continuing low return rates of adults that spawn in the Snake River. This unexplained mortality of Snake River salmonids outside the migration corridor is called "extra mortality". Many hypotheses for the cause of this extra mortality have been proposed: the hydrosystem itself may weaken fish and disrupt their natural rhythms, hatcheries may interfere with the fitness and survival of wild fish, habitat degradation may have reduced stock vigor, genetic effects may have reduced stock viability, and the degraded ocean conditions may have differentially taken a toll on salmonids that spawn above the Snake River dams. The impact of dam breaching compared to keeping the dams intact depends on which of these alternative sources of extra mortality one assumes to reflect the "truth". Statistically and quantitatively sorting through these uncertainties has been a major task for PATH as well as for the assessment undertaken by NMFS. New PIT-tag technology, the emergence of large-scale geographic data bases, and experiments with hatcheries provide opportunities for science to address uncertainty about extra mortality on the time scale of 10 to 20 years.

1.3 Species-by-Species Accounts

1.3.1 Spring/Summer Chinook Salmon

PATH results indicate that breaching, under a wide variety of assumptions, is more likely to meet recovery and survival criteria for spring/summer chinook salmon than transportation options. Although it is clear from PATH results that transportation options are less likely to meet the survival and recovery criteria, how much less likely depends critically on assumptions about the extent to which transported fish suffer a differential delayed mortality in combination with assumptions about the sources of extra mortality not explained by transportation (hydrosystem, regime shift in the ocean, or reduced stock viability).

The analytical results for spring/summer chinook salmon exemplify the key uncertainties that surround any assessment of the impact of management actions on salmonids in the Snake River Basin. Specifically, different fish passage models use contrasting estimates of the extent of differential delayed transportation mortality. Moreover, recent measures of differential delayed transportation mortality appear to be lower (higher D-values) than estimates from the past. Simulations show that, if one uses high D-values (which mean that transportation does not cause substantial delayed mortality), then the mean advantage of dam breaching over current operations across all sets of assumption regarding extra mortality is reduced from 30% to 11% (using estimates from the recent two years of empirical data as the defining D-value). In addition, if this transportation mortality is as low as the most recent estimates, then the average advantage of breaching can fall to as low as 2% for certain assumptions about extra mortality. This highlights the value of resolving uncertainty about transportation mortality in conjunction with sources of extra mortality.

The bottom line is that, under current levels of uncertainty, breaching meets recovery and survival goals over the widest range and highest proportion of assumptions for spring/summer chinook salmon. However, this "clear message" is somewhat muddied by the uncertainty about differential delayed transportation mortality. The options for managers are to pursue breaching immediately (a risk-averse strategy) or to delay breaching as more information is gathered to resolve critical uncertainties. There is a cost to delaying. That cost can be measured as any additional probability of failing to exceed survival thresholds, on average, because breaching is not initiated. According to the PATH simulations to date, that additional risk with respect to survival standards is 8%. However, the actual cost of this delay is likely to be higher because PATH models do not represent extinction risks (i.e., the models do not include any scenarios involving stocks becoming extinct), and many of the spring/summer chinook salmon stocks are currently so low that extinction risk is substantial over the short term.

1.3.2 Fall Chinook Salmon

PATH results for fall chinook salmon parallel those for spring/summer chinook salmon, but are extremely preliminary. Drawdown is more likely to meet recovery criteria under most assumptions and uncertainties, whereas transportation is not. Unlike the situation with spring/summer chinook salmon, uncertainty about differential delayed transportation mortality (D-values) is less critical to the evaluation of management options for fall chinook. There is an important additional route by which drawdown is expected to benefit fall chinook salmon, even without assuming that there is a differential delayed mortality due to transportation. In particular, because fall chinook salmon spawn in the mainstem river as opposed to tributaries and streams, drawdown is expected to increase the carrying capacity (available habitat) for fall chinook salmon by more than 70%. This increase in carrying capacity is independent of any assumptions about transportation mortality, but could not be achieved without drawdown.

1.3.3 Steelhead

We do not have quantitative recovery and survival criteria for Snake River steelhead. In addition, there are insufficient data to produce quantitative analyses at the level of detail possible for chinook salmon. The one logical framework that is possible given this paucity of data is to explore the extent to which steelhead “behave like” chinook salmon, and then to use results from chinook salmon to draw conclusions about steelhead. This has to be done with caution, because although there are aspects of steelhead population trends that mirror spring/summer chinook salmon, there are also notable discrepancies. In general, whatever “works” for spring/summer chinook salmon is likely to work for steelhead. However, management that does not sufficiently recover chinook salmon could still recover steelhead stocks.

1.3.4 Sockeye Salmon

The absence of data is even worse for sockeye salmon in the Snake River than it is for steelhead. Currently, a captive brood stock program maintains the sockeye salmon population in the Snake River. Numbers are so low, that there are no prospects for generating life-cycle data of the caliber needed for a formal risk analysis or recovery and survival analysis. Some information about factors such as ocean conditions and hatchery effects on sockeye salmon may be gleaned from studies of sockeye salmon in the mid-Columbia reach (but the hydrosystem effects in that reach are not comparable to the Snake River hydrosystem effects).

1.4 An Ecosystem Perspective

All four of the listed salmonid species exist in a complex ecosystem, with a wide variety of threats and factors that determine their biological fates. The PATH process has focused on hydrosystem actions and their impacts, and has dealt with broader threats primarily by performing sensitivity analyses. For example, the future is simulated with different schedules of harvest intensity or different temporal patterns of ocean conditions. It is important to ask how such a broader view changes the species-by-species conclusions enumerated above, and how this broader view could alter conclusions about the relative merits of hydrosystem management actions. Without belaboring details, common sense makes it obvious that if actions outside the hydrosystem can dramatically improve survival rates or productivity for listed species, then dam breaching may not be necessary. For example, if hatchery production were dramatically curtailed, some mortality currently suffered by wild stocks might be reduced and, in conjunction with transportation, might ensure survival and recovery. Alternatively, if ocean conditions dramatically improved, all stocks could fare much better. What this means is that there are suites of scenarios that could lead to recovery without breaching. But conversely, there are also numerous suites of conditions corresponding to a deteriorating situation for the listed species that might make it less likely that breaching alone could meet recovery and survival criteria. Examples of such pessimistic possibilities include degradation of stock viability due to the erosion of genetic variability and ocean conditions that remain poor (or even get worse). These broader ecosystem perspectives point to three major general areas of study demanding further attention:

- The connection between habitat quality and productivity (since land use patterns, hydrosystem actions and pollution all influence habitat quality);

- The biological mechanisms underlying the linkages between ocean conditions and salmon adult survival and growth (here multispecies studies and ecosystem analyses will be necessary); and

- Greater attention to hatchery impacts, potentially through adaptive management experiments.

The NMFS has already launched research initiatives to address questions about changing ocean conditions and their impact, as well as questions about improved hatchery operations and the connection between habitat conditions and salmon productivity. However, these are long-term investigations that will reap rewards on the timescale of decades, not a few years. The above three research areas address the uncertainty about extra mortality, an area of uncertainty that PATH indicates is crucial to the assessment of how well different management options will perform. There is some chance that the knowledge gained from such research could identify management strategies in which breaching is not necessary for recovery and survival. However, it is reasonable to conclude there is an equal probability that the lessons from broader ecosystem studies will reveal that breaching is only one of several ecosystem management actions that are needed for salmon recovery, and is thus a necessary first step.

1.5 Conclusions

1. Breaching is more likely than any other hydrosystem action to meet survival and recovery criteria for the listed species across the widest range of assumptions and scenarios. Thus breaching is the most risk averse option.

2. However, there are plausible sets of assumptions under which breaching yields little or no improvement over transportation alone – most notably if differential delayed transportation mortality is assumed to be low for spring/summer chinook salmon, the advantages of dam breaching are not as compelling. This points to the value of narrowing our uncertainty about transportation mortality from PIT-tagging studies that have already been initiated by NMFS with spring/summer chinook salmon.

3. Learning more entails delays, and delays entail risks. There would be a 5-10 year delay associated with gathering additional information, the "answer" still will not be certain, and there is some risk that if data reveal transportation mortality is in fact substantial, the species suffer an enhanced risk of not meeting the survival criterion. That risk, according to PATH analyses to date, is an approximately 8 % greater chance of failing to exceed the survival escapement threshold. The actual risks are likely to be higher than estimated by PATH because some localized stocks are so small that extinction is a possibility over the short term. Indeed some stocks are so low, that every action, even those actions that in the framework of PATH analyses seem to offer only small benefits, could be critical to getting those stocks through low bottlenecks to population levels where they are less vulnerable to chance misfortune.

4. Assessments to date have focused on hydrosystem effects, with some simulations or sensitivity analyses having been conducted to look at other factors such as harvest, hatcheries, and habitat. However, the potential improvements in salmonid stocks as a result of restored habitats or improved hatchery practices have not been explored in detail. The NMFS has initiated a research program for examining these other risk factors in one common framework, so that clearer answers can be given about what management is necessary or sufficient for salmonid survival and recovery.

5. In summary, the NMFS concludes that breaching is more likely than any other hydrosystem action to promote recovery under the widest range of assumptions – in other words breaching is the most risk-averse strategy. However, while it is clear that breaching is the most risk averse option, it is not certain that breaching is absolutely necessary — that depends on which assumptions are correct. There is no simple answer – there are only trade-offs between potential risks and benefits to be weighed and considered. The weighing of these trade-offs depends largely on the extent to which "risk averseness" is favored – choosing a level of risk averseness is a policy question and hence is beyond the scope of this report. Scientific data can be used to estimate risks, but only policy can decide what level of risk is acceptable given the many other constraints surrounding such important decisions.

2.0 Introduction

2.1 The Need to Examine Management Options

Salmon populations in the Snake River have been listed under provisions of the U.S. Endangered Species Act (ESA). The pertinent listed species are: Snake River sockeye salmon (*Oncorhynchus nerka*, listed as Endangered in 1991), Snake River spring/summer and fall chinook salmon (*O. tshawytscha*, both listed as Threatened in 1992), and Snake River steelhead (*O. mykiss*, listed as Threatened in 1998). Because of these listings, there is a need to consider management options that might mitigate the threats to these populations and assist in their recovery. This report focuses on an ecological assessment of management alternatives for the Federal Columbia River Power System (FCRPS).

The National Marine Fisheries Services' (NMFS) 1995 FCRPS Biological Opinion (NMFS 1995) concluded that major changes were needed to significantly increase salmon survival. NMFS called for a detailed evaluation of alternative configurations and operations of the four Federal hydroelectric projects on the lower Snake River. The purpose of this evaluation is to determine the likelihood that drawdown (breaching) of these four projects, or some other alternative such as expansion of the juvenile fish transportation program, would result in the survival and recovery of Snake River salmon and steelhead. In support of its Lower Snake River Juvenile Salmonid Migration Feasibility Study (Feasibility Study), the U.S. Army Corps of Engineers (Corps) requested that the National Marine Fisheries Service summarize available information on the potential effects of the management options on anadromous salmon and steelhead runs originating within the Snake River system. This report responds to the Corps' request. Because the effect of any hydrosystem action would be embedded in the broader relationship between fish and their environment, management actions are evaluated in the context of factors that might occur outside the direct control of the hydrosystem (such as hatcheries output and changes in habitat, harvest, and ocean conditions). The science of ecosystem management is still in its infancy; although the value of such an ecosystem approach is widely appreciated, scientists are grappling with how to implement this viewpoint in practice.

2.2 Preview of the Document

The report begins with a discussion of the general analytical framework that guided the ecological assessment of management options. It is important to understand the logic underlying the analyses. Before turning to each of the four salmonid species of interest (spring/summer chinook salmon, fall chinook salmon, steelhead, and sockeye salmon), this report reviews features of the historical trends for salmonids, in general, in the Snake River and basic salmonid life history attributes. Then each species is taken up in turn. There are more data for spring/summer chinook salmon than for fall chinook salmon, and much more than for sockeye salmon or steelhead. Hence the level of detail in this report is uneven among species. All of the species-specific discussions are, however, subdivided into the same key five sections:

- Historical trends,

- Adult harvest and upstream passage,

- Egg-to-smolt life stage,
- Smolt-to-adult life stage, and
- Analysis of hydrosystem management alternatives.

The species-by-species discussions are followed by a general discussion of experimental management options and the opportunities for learning more in the near future. No matter which management option is pursued, it will require a carefully designed monitoring and evaluation plan so that outcomes are linked to the action.

3.0 Analytical Approaches Used to Evaluate Management Options

3.1 Relationship Between PATH Process and NMFS Report

This anadromous fish assessment is a product of the NMFS Northwest Fisheries Science Center (NWFSC). In developing this report, we rely extensively on the syntheses and analyses conducted in the regional process known as the **P**lan for **A**nalyzing and **T**esting **H**ypotheses (PATH, described in **Section 3.2**). As a component of the NMFS Regional Forum for implementing the FCRPS Biological Opinion, the PATH process has quantitatively examined the biological consequences of alternative hydrosystem actions, and those results are generally pertinent to the issues addressed in this report. Although this report draws on the results from PATH, it has not gone through the PATH process. Wherever results are taken from PATH documents, those documents are referenced; however, scientists for the NWFSC have independently reviewed the analyses of the PATH process and synthesized those results to produce NMFS' conclusions.

3.2 Logical Framework of Analyses

It is very difficult to develop simple recipes for management that are well grounded in clearcut scientific data. Within their complex life histories, salmon and steelhead are exposed to many factors that influence their ultimate prospects for survival and recovery. This report approaches the challenge of assessing the likely effects of manipulating the hydrosystem by using a multivariate statistical analysis tailored to the complexity of the problem. Specifically, the report breaks the salmon life cycle into stages and imposes on these stages a variety of assumptions about baseline conditions and likely changes due to different management actions. Historical data are used to narrow the range of assumptions and establish the magnitude of uncertainty; life-cycle models are then used for two of the species (spring/summer and fall chinook salmon) to project the "likely effects" of actions into the future. Inferences from these detailed analyses and from the scientific literature are used to draw conclusions for the other two species (steelhead and sockeye salmon) for which few data exist.

A decision analysis was used to tackle the complexity and uncertainty of salmonid survival and recovery. This analysis was quantitative for the two chinook species, but more qualitative for steelhead and sockeye salmon. The five steps in this analysis were:

- Specifying an array of assumptions and uncertainties based on historical data;
- Embedding the above assumptions in models that project futures under different management options and scenarios;
- Summarizing these predictions of "potential futures" in terms of the likelihood of meeting survival and recovery criteria (i.e., populations are intended to be above minimum abundance levels [survival] and to even increase to higher abundance levels [recovery]);
- Identifying the critical uncertainties that have the greatest impact on the predictions; and
- Synthesizing the results and sensitivity analyses into summary statements about the biological merits of alternative management options.

As mentioned in **Section 3.1**, much of the decision analysis was provided by the PATH process. It is useful to quote exactly the words from a PATH report (page 1 in Marmorek et al. 1998a) to describe the 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 reduce uncertainties in the fundamental biological issues surrounding recovery of endangered spring/summer chinook salmon, fall chinook salmon, steelhead and sockeye salmon 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)."*

Before turning to specifics, it is worth reviewing the general logic underlying the PATH process. PATH summarizes the results of applying a detailed life-cycle model to predict future chinook salmon populations under a variety of management alternatives. To implement the model, eight to ten different key assumptions are required (i.e., depending on the species examined, with most of the assumptions corresponding to a specific "rate" or "parameter" in the model). A great deal of work went into defining all of the critical assumptions and the uncertainties that underlie them. However, it is important to realize that PATH is not locked into a rigid set of assumptions – as new ideas are generated, PATH can run new simulations with new assumptions. This flexibility and openness to participant input (where participants are Federal, state, and tribal resource agencies, and independent scientists) is one of the strengths of PATH. On the other hand, in synthesizing this report, NMFS has concluded that PATH's openness needs to be balanced with an increased commitment to discarding those hypotheses with weak empirical support, or combining hypotheses that do not noticeably affect the results. By narrowing the set of assumptions under which models are run, the comparison of management options will become more transparent.

To fully evaluate the likely effects of management actions on chinook salmon, PATH simulations were run under a wide variety of assumption sets. The word "run" refers to one particular set of assumptions. For each run, 4,000 replicate Monte Carlo simulations were executed. Thus, each run actually produced 4,000 different projections into the future (reflecting the reality that environmental variability requires that futures be represented as frequency distributions of likely outcomes rather than a single deterministic result). For each management action, a large number (**ranging from 240 to 1,920**) of different assumption sets or "runs" were examined. Recently, the PATH process initiated a procedure for narrowing some of the uncertainty associated with salmon life-cycle modeling. In particular, PATH convened a panel of four experts (called the Science Review Panel or SRP) in which the panel was asked to weight alternative assumptions for each of seven different hypotheses that are required to feed into the life-cycle modeling and future simulations.

In this report, NMFS does not use the results from SRP-weighted assumptions for three reasons: (1) clarity, (2) using the weighted assumptions does not qualitatively alter any of the conclusions (Marmorek et al. 1998a), and (3) new data render some of the weightings obsolete. In particular, new data are becoming available which will allow us to reject alternative hypotheses via standard statistical methods as opposed to using expert panels.

In noting this difference between PATH and NMFS with respect to weighted assumptions it is useful to put PATH in a broader context than simply the formal self-description of its goals quoted above. PATH was born in 1994 out of the vision that rather than unproductively and relentlessly engaging in arguments about different models and different hypotheses about the Columbia Basin salmon stocks, all of the different perspectives ought to be brought together in one group for a common analysis and decision-making framework (Marmorek et al. 1996). PATH coordinates and reviews alternative life-cycle and passage models or analyses so that they at least share a common reporting terminology and currency; but PATH does not conduct primary research. Despite four years of working together, there remain fundamental disagreements among PATH participants about crucial hypotheses (see **Section 4.4**). PATH has focused primarily on the hydrosystem, and has not yet had sufficient time to complete all of the analyses requested by participants. PATH has also been a learning process. It is disingenuous and unfair to criticize PATH for "mistakes" that are only clear with the hindsight of analyses conducted by PATH itself. PATH has made extremely valuable contributions, yet, with the benefit of everything we now know, there are results from PATH (such as the SRP weighted hypothesis analyses) and aspects of PATH upon which NMFS does not draw when examining the likely effects of alternative hydrosystem actions. Even though NMFS has participated in PATH, it is natural that NMFS is constantly updating its own scientific views with new information. Naturally, NMFS is submitting this report to PATH as a distillation of the latest NMFS thinking.

3.2.1 Developing Performance Measures

The performance measures used to judge the adequacy of the modeled alternatives were those used by NMFS along with non-quantitative considerations put forth in the 1995 FCRPS Biological Opinion. These performance measures were criteria for the survival and recovery of listed stocks. Clearly, the complexity of results entailed in the simulated (i.e., modeled) projections requires some form of synthesis before it is useful to fisheries managers. Therefore, each set of model runs was summarized relative to these two performance criteria.

The performance criteria used to assess the likelihood that a stock would survive is the requirement that 70% of years the spawning abundance of a stock is above a certain low threshold within the 24- or 100-year time period. The specific threshold level depends on the characteristics of each stock and its natal stream (BRWG 1994 and Appendix D in Marmorek and Peters 1998a). One important caveat is that the PATH prospective models did not explicitly evaluate the risk of absolute extinction. Yet, with the low numbers of some of the simulated stocks, extinction is a reasonable possibility (Lande 1993). Some analyses suggest, however, that the failure to include extinction does substantially affect forecasts of the ability to meet survival and recovery criteria (Botsford 1997). Nonetheless, the absence of an extinction analysis suggests that conclusions about the likelihood of satisfying survival criteria are optimistic. In other words, it would reduce prospects for survival and recovery of salmonids would if extinction were allowed in the prospective simulations of future population trends.

The probability of meeting the survival criterion under a particular set of assumptions is the fraction of 4,000 replicate Monte Carlo simulations that result in an average abundance of spawners exceeding their survival threshold population level for 70% of the years. PATH examined survival criteria for 24-year and 100-year timeframes. In contrast, the chance that a certain hydrosystem action will recover a spawning stock is described by the fraction of 4,000 replicate simulations for which the average spawner abundance over the last 8 years of a 48-year simulation is greater than a specified level (BRWG 1994). A recovery level was assigned for each index stock based on historical census data. In particular, each stock's recovery level was set to be 60% of the average spawner counts from before the 1971 brood year.

To determine whether this target had been reached, PATH and NMFS apply a geometric rather than arithmetic mean to prospective simulated populations. In contrast to a straightforward arithmetic mean, a geometric mean is reduced in proportion to variability in year-to-year population counts. Thus, the arithmetic and geometric means of 100, 100, and 100 are the same – 100; whereas the geometric mean of 1, 100, and 199 is only 27 (compared to an arithmetic mean of 100). This discounting for variability is well-founded in population biology, because sustainable harvest is diminished by population variability (Lande 1997). The actual recovery criterion that NMFS focuses on for each stock requires that the geometric mean population size over the last 8 years of the simulation exceeded the "60% target" mentioned above.

The probability associated with each performance criterion for a given action can be compared with any likelihood level considered critical to a decision by managers. Estimating the magnitude and probability of effects is a scientific process (risk analysis), whereas determining the acceptability of risks (risk management) is not (Suter 1993). The maximum acceptable risk level (that is, the minimum acceptable likelihood of meeting performance criteria) is specific to the particular management decision under consideration and will rely largely on the legal and policy context of the decision. Because management criteria are currently unknown, PATH displayed performance measures in a manner that allows comparison with any likelihood deemed acceptable by a given management entity.

The PATH report, at the suggestion of NMFS' PATH members, also identified probabilities that roughly approximate probabilities associated with sets of actions determined not to jeopardize listed species in the 1995 FCRPS Biological Opinion (NMFS 1995). The NMFS has articulated a qualitative survival criterion requiring that a "high percentage" of available populations must have a "high likelihood", of meeting these survival criteria over each time period and has defined "high percentage" as 80% of available populations (NMFS 1995). However, the level of 80% does not neatly transfer into a specific number of stocks in the case of the seven index stocks for Snake River spring/summer chinook salmon (see **Section 5.1)**. Five index stocks would comprise 71% of the available populations and six index stocks would comprise 86%. Therefore, PATH assumed that six of the seven stocks should have a high likelihood of exceeding the threshold number of spawners over time. NMFS did not define "high likelihood" but PATH assumed, based on NMFS' PATH members' recommendations and Tables 3 through 6 in NMFS (1995), that a simulation would satisfy NMFS' survival criterion if six of the seven stocks were above a stock-specific threshold at least 70% of the assumption sets. Similarly, NMFS qualitative recovery criterion states that a "high percentage" of available populations must have a "moderate to high likelihood" of exceeding these recovery thresholds. For the same reasons described above, PATH assumed that "moderate to high likelihood" was achieved if six of the seven stocks were above a stock-specific threshold 50% of the assumption sets.

In summary, the relative likelihood that a listed species will survive and recover under alternative configurations (e.g., dams out; maximum transportation) is described in two ways.:

1. By the average fraction of replicate Monte Carlos simulations (averaged across all assumption sets) that meet threshold population escapement levels and recovery levels (usually reported in tables such as **Table 5.5.1.3-1**).

2. By the distribution of probabilities for meeting survival and recovery thresholds, which can then be compared to particular risk thresholds (usually presented in "Box and Whisker" graphs such as **Figure 5.5.1.3-1**).

The PATH analyses used 24-year and 100-year survival criteria and 48-year and 100-year recovery criteria. The NMFS examines the alternative hydrosystem actions in terms of the 24-year survival and 48-year recovery criteria. There are two reasons for selecting these two out of four possible performance measures.

- The 48-year recovery criterion is the criterion that provides the greatest distinction among management actions. The other criteria are important; however, when trying to choose among alternative actions, the measures that generate the sharpest contrast in relative probabilities offer the most guidance.

- The 24-year survival criterion is the shortest time scale over which any quantitative analyses were performed. Thus, the survival criterion can help measure the cost of not taking action over the short run.

A good biological way of thinking about these criteria is that the 24-year survival thresholds measure vulnerability to extinction over the short run whereas the 48-year recovery thresholds emphasize rebuilding populations to higher levels.

It is important to note that, although particular numbers were chosen for these survival and recovery criteria, when it comes to comparing management actions, <u>all comparisons are relative</u>. Thus, raising or lowering a threshold might change the absolute frequency of runs for which criteria are satisfied, but the differences between management actions (in terms of relative probability) is much less sensitive to the specific threshold level chosen.

There is currently vigorous scientific debate about the ability of science to predict population survival (or conversely, extinction) because catastrophes are so unpredictable (Ludwig 1999). In contrast to survival thresholds (for which rare chance events can override average behavior), the behavior of populations near recovery thresholds are less vulnerable to chance and more predictable. As a decision tool for comparing management, NMFS has decided to focus on recovery for two scientific reasons:

- Recovery is probably more tractable using analytical methods and

- Focusing on recovery is a risk-averse approach, because if one achieves recovery, survival is much more effectively guaranteed.

There is so much uncertainty and unpredictability surrounding the population dynamics of most threatened or endangered populations, that risk-averse decision-making is widely recommended (National Research Council 1996a). It is important to note that for this anadromous fish assessment and related PATH analyses, using criteria other than the 48-year recovery measure does not change the relative differences among alternative management actions; it merely changes the relative likelihood that all actions meet the threshold criteria under consideration. Survival criteria are brought into the analysis largely to quantify the risk of delaying actions.

One way of summarizing the myriad results from PATH is to simply calculate the <u>average</u> fraction of simulations that satisfy a survival or recovery criterion across all of the assumption sets. The PATH documents refer to this as an "average probability" (see Marmorek et al. 1998a) of meeting survival or recovery criteria. NMFS thinks it is important to avoid referring to these average fractions (or percentages) as probabilities – instead they represent <u>relative probabilities</u>. NMFS emphasizes <u>relative</u> because the fraction or percentage always depends on the size of the set of assumptions used to generate the simulated projections, rather than some absolute reality. The more alternative assumptions that are included, the smaller the weight assigned to any one assumption when all are weighted equally. Thus, the probability is partially determined by the number of alternative assumptions under consideration. For example, consider the fact that 240 assumption sets were used to model the future for the status quo (i.e., alternative A1, the existing condition). If one were to decide that one additional assumption (with two possible values) should be considered, then suddenly there would be 480 (= 2 x 240) assumption sets. What looked like a probability of 70% for the 240 assumption sets could change to anything from 35% ([0.7 x 240 + 0 x 240]/2) to 85% ([0.7 x 240 + 1.0 x 240]/2). This is not a trivial point. These "relative probabilities" do not translate in any way to a true probability (in the sense that we know the probability of getting a "heads" when we flip an honest coin is 0.5). True probabilities are possible only if we are absolutely certain about the "true" number of critical assumptions and the "true" definition of the alternative states of each critical assumption. The practical point is that the "relative probabilities" as defined by the PATH process do not represent the "true probabilities" intended when making a jeopardy decision. The relative probabilities are useful for comparing the <u>relative</u> merits of different management options with respect to survival and recovery – but they are not literal probabilities regarding the fate of the populations. <u>The predictions generated by the PATH analyses do not provide absolute predictions and should not be misinterpreted as such.</u>

A second major way that PATH summarized and interpreted its results across all assumption sets was by identifying those management options that are "most robust" – in other words those management options that work under the widest range of assumption sets. Clearly if we believe all assumption sets are equally likely and if a particular management option achieves success (as measured by the above performance measures) for 100% of the possible assumption sets under consideration, that management action has something to recommend it above a management alternative that achieves success for only 60% of the assumption sets under consideration. Moreover, by identifying those assumption sets that do not yield success under certain management scenarios, we learn what uncertainty requires resolution in order for us to have confidence that a management action would succeed.

In a perfect world, science would remove a large portion of the uncertainty inherent in both approaches, and fisheries managers would not face a complicated array of hundreds or thousands of different hypotheses. However, uncertainties exist and scientists must respond to the immediate need to support resource management decisions while continuing to develop better analyses and gathering more data. There is no menu or handbook of how to incorporate uncertainty into decision analyses. Hence, this report addresses uncertainty in three ways:

- Via a presentation of the relative probabilities derived by PATH;
- Using sensitivity analyses to explore different future scenarios (also a part of the PATH process); and
- Through a more general discussion of the biological sources of the uncertainty, and how the alternative hypotheses might be tested.

3.3 Basic Field Data Used for Run Reconstructions: Quality Control and Quality Assurance

The primary data upon which all of the run reconstructions, and hence the retrospective analyses of stock performance, are based consist of spawning redd (or "nest") counts. For some index stocks, redds were counted only over a portion of a creek's length and were then extrapolated to derive a count for the entire length of the creek. The annual number of spawners is then calculated by multiplying the number of redds by the estimated number of fish per redd (Beamesderfer et al. 1998). There are several potential sources of error in field counts of spawning redds. First, as with any field sampling program, there may be straightforward observation errors (redds might be missed or mistakenly double counted). In addition, sampling error may occur because the methods for sampling vary – sometimes taking the form of aerial surveys and other times the form of ground counts. Of the two methods, it is more likely that the accuracy of aerial surveys is influenced by weather. Another source of error is the timing of redd counts – if censused too early, the number of redds would probably be underestimated. Finally, the fact that different observers are used introduces the potential for observer bias, with the possibility of "learning" creating temporal trends in an individual's "bias". Petrosky (1996) used correlations between the number of redds counted and the number of spawners counted at weirs to estimate the magnitude of error in redd counts and found an r-squared value of 0.91 and a 24% coefficient of variation for the ratio of redds counted to female escapement. Unfortunately, this estimate of error was performed for stocks in the Lemhi, Upper Salmon, and Crooked rivers, none of which correspond to the actual index stocks used in the PATH analyses. Because survey data contribute to adaptive management decisions, greater attention should be paid to estimating the magnitude of error in the future collection of primary data for the index stocks. NMFS has recently initiated basic research on monitoring programs for salmonids, so that critical levels of observation error might be identified for different questions and sampling designs.

Nonetheless, one can examine how the Petrosky (1996) estimate of observation error the run reconstruction methodology. Deriso et al. (1996) found that a 25% coefficient of variation did not markedly alter the PATH life-cycle model's ability to estimate total passage mortality. It would be useful to broaden these assessments of error propagation to include larger observation errors and to also consider the impact of potentially anomalous years on model performance. Because the PATH quantitative approach emphasized the "risk averse" perspective applied to a wide range of hypotheses and scenarios, these issues of data quality and control were not as important as if the data were used to directly inform decisions. However, as NMFS proceeds to narrow down the range of hypotheses, data quality and control will become increasingly important.

3.4 Defining the Management Options

The basic intent of the anadromous fish assessment is to provide a summary of available biological information pertinent to the effects of the various Snake River hydrosystem management alternatives under consideration in the Feasibility Study. Evaluating the potential response of Snake River salmon runs to the alternative hydrosystem configurations requires consideration of the population dynamics of the Snake River stocks, direct and indirect impacts of each action on adult and juvenile survival, future climate and environmental impacts, and the effects of harvest, hatchery, and habitat actions or strategies. The PATH process has examined in varying degrees the seven different management options listed below (and summarized in **Table 3.4-1**):

- 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 passage option (no transportation, no drawdown, flow augmentation as in A1, plus 1 million acre-feet from upper Snake River, and surface bypass

systems). This option has not yet been fully developed, so PATH performed a preliminary qualitative assessment of its probable effects on spring/summer chinook, relative to the other actions. A similar analysis for fall chinook salmon is planned but not yet completed

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

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

Other options, such as drawdown without flow augmentation, were not quantitatively analyzed, but are discussed in the draft Fish and Wildlife Coordination Act Report (USFWS 1998).

Analyses of these different options by PATH vary in detail. This report focuses primarily on contrasting option A3 (drawdown of four Snake River dams) with option A1 (essentially the current system, with transportation of fish) or with A2 and A2' (existing system with structural improvements). The effect of these management options was generally examined under a variety of scenarios (such as alternative harvest rates) as well as across a wide range of assumptions.

3.5 Needs for More General Risk Analysis

It is important to reiterate that there are limitations to the PATH approach, which entailed exhaustive simulations of thousands of different, equally likely assumptions. First, the analyses are only as thorough as the extent to which all plausible assumptions are thoroughly delineated. Second, the PATH approach is an important first step toward resolution of major uncertainties, but not the only step. By making explicit the connection between particular sets of assumptions and the outcome of management actions, it is possible to strategically explore and identify research opportunities aimed at testing the most critical assumptions. Thirdly, the large number of “assumption sets” currently analyzed is surely too broad, and will be narrowed with further research. Fourth, “robustness” is not the only logical framework for evaluating management options. For instance, a reasonable alternative to seeking the management option that works best regardless of what we believe is true (the “most robust”), is to seek the management option that works best given the single most likely set of assumptions. In other words, instead of seeking the “safest” management option (works no matter what is assumed), another sensible approach seeks the “best” option (works best under the single assumption set that seems most likely to be true). Seeking the management option that works best under the single most likely set of assumptions carries with it a risk of “being wrong” about which assumption set is the “correct one”. In general, risk analyses seek to balance attention to robustness (risk aversion) with efficiency (identifying the single best answer) (Morgan and Henrion 1990). Finally, because PATH is an ongoing process, it has not yet itself pursued all of the viable comparisons among assumption sets. Therefore, in preparing this report an effort was made to identify issues not yet explicitly addressed via PATH (to date) and also to identify potential surprises that do not readily fit into the analytical framework developed by PATH. In this sense, the report that follows is a broader risk analysis than that provided by PATH alone.

In summary, the analytical framework used by both NMFS and the PATH process draws on historical data to systematically specify what we currently know and do not know pertinent to the survival and recovery of salmon and steelhead from the lower Snake River. The point of this analysis is to compare hydrosystem management scenarios with respect to which options meet certain performance criteria under the broadest array of different assumptions. This quantification of “robustness” provides a basis to recommend particular immediate actions, and to identify the uncertainties that need resolution before managers can feel confident that they understand all the implications of choosing a particular management option.

This NMFS report is “hydrocentric” in the sense that it focuses on risks to salmonid populations posed by the hydrosystem, and how modifications of the hydrosystem could foster the survival and recovery of threatened populations. It is well known, however, that the hydrosystem, is only one of several factors that have contributed to the decline of salmon populations (NRC 1996b). Nonetheless, a hydrocentric focus for this Appendix is warranted for two reasons:

- The Anadromous Fish Appendix is a critical component of the Lower Snake River Juvenile Salmonid Migration Feasibility Study, which was required by NMFS as an element of the Reasonable and Prudent Alternative in the 1995 FCRPS Biological Opinion. The objective of the assessment is to evaluate the scientific evidence of the likely effects of alternative hydrosystem actions on listed Snake River salmon and steelhead.

- Because so much research has recently been directed at patterns of survival for juvenile salmonids passing through mainstem hydrosystem projects via different routes (spill, barge transportation, juvenile bypass systems, and so on), the best quantitative data relating risk factors to salmonid demography pertain to hydrosystem factors.

NMFS will pursue analyses that are not so “hydrocentric” as it develops recovery plans for the threatened salmonid stocks. However, it is important to note that other factors affecting Snake River salmonids (hatcheries, habitat, genetic alterations, etc.) ARE NOT neglected in this Appendix because of the hydrosystem focus. Instead they are generally “lumped together” under broad categories of “extra mortality” (see **Section 4.4**), or are examined in the context of sensitivity analyses that play out a handfull of alternative scenarios for harvest schedules and ocean conditions. An instructive analogy relates to a decision of whether to perform bypass surgery on a patient as opposed to a combination of numerous other therapies (altered diet, drug therapy, angioplasty, and so on). The initial consideration of surgery would focus on whether or not to perform the “bypass” operation. Similarly, this assessment focuses on “dam breaching” versus “no breaching”. However, this focus does not mean that other risks to salmon are not considered – these other risks enter the life-cycle model through the general mortality terms and through the term for “extra mortality” described in **Section 4.3**. But, instead of separating out marine mammals and bird predation as individual risk factors, they are lumped together as “*other*” risk factors, where “other” refers to “*other than hydrosystem*”.

In addition, NMFS is in the process of initiating a cumulative risk analysis that places all of the risks on equal footing and with a common or comparable demographic currency (see **Section 10.5**). This broader risk analysis will necessarily be pursued at a much cruder quantitative level than was possible for the hydrosystem actions examined by PATH and NMFS in this report – this is because the data for the other risk factors cannot be as easily associated with discrete steps in a salmonid life-cycle model. For the dam breaching analysis requested by the U.S. Corps of Engineers, the level of detail applied to hydrosystem factors is both possible and appropriate.

4.0 An Overview of the History and the Life-Cycle of Salmon in the Snake River

4.1 Historical Trends

The Snake River historically was and presently is one of the most important drainages in the Columbia River system for producing salmon. More broadly, salmon in the entire Columbia River system at one time numbered between 10 and 16 million fish; this drainage once contained the largest chinook salmon population in the world. Estimating specific historical population levels and trends of particular stocks of salmon in the Snake River subbasin of the Columbia River is more difficult. But it is clear that all salmonid stocks in the Snake River were much more abundant at the end of the nineteenth century than they are now and that these stocks have undergone major fluctuations. Before turning to detailed accounts of spring/summer chinook salmon, fall chinook salmon, steelhead, and sockeye salmon, it is worth reviewing general trends and basic common life-history stages.

Declines in Columbia River salmon populations from peaks in the 1880s began at the end of the last century as a result of overfishing; however, by early in the 20th century, environmental degradation from mining, grazing, logging, and agriculture caused further declines. Prior to construction of the first mainstem hydroelectric dams on the lower Columbia River (Bonneville Dam was completed in 1938), aggregate pounds of chinook salmon (*O. tshawytscha*) caught in the Columbia River had declined by approximately 40% since the beginning of the century (Netboy 1974). More recent historical decreases in Snake River stocks coincided with an intensive period of change from 1953 to 1975 in the middle and lower Snake River and the lower Columbia River. In addition to construction of the impassible Hells Canyon complex of dams, four dams which allowed varying degrees of passage were built in the lower Snake River and three in the lower Columbia River. The completion years during this period were 1954 (McNary Dam), 1957 (The Dalles Dam), 1958 (Brownlee Dam), 1961 (Ice Harbor and Oxbow Dams), 1967 (Hells Canyon Dam), 1968 (John Day Dam), 1969 (Lower Monumental Dam), 1970 (Little Goose Dam), and 1975 (Lower Granite Dam). The seven new dams on the lower Snake and Columbia rivers inundated 227 and 294 kilometers of mainstem habitat, respectively. This changed the lower mainstem river from a mostly free-flowing body into a series of reservoirs covering about 70% of the distance between Lewiston, Idaho, and the Pacific Ocean. The slow-moving reservoirs decreased the rate of downstream travel for juvenile fish and increased the amount of habitat favorable to occupation by exotic and predator species. The construction of new dams was one of a suite of major changes in the Columbia basin ecosystem. Other major changes that had potentially significant impacts on salmonid populations included: the emergence of industrial-scale hatchery production, the introduction of exotic species, major shifts in oceanic conditions, and dramatic seasonal shifts in water storage and flow regulation (NRC 1996b). In **Section 4.4**, the technical issues surrounding an analysis of hydrosystem effects in the context of these many other risk factors are discussed.

4.2 General Life Cycle of Snake River Salmon

The salmon life cycle provides a framework within which to assess the factors leading to the decline of Snake River salmon runs and to evaluate the potential impact of alternative actions aimed at salmon protection and recovery. Human activities can affect survival during each major phase of the life cycle (NRC 1996b).

4.2.1 Adult Stage

Salmon originating in the Snake River reside in the ocean for a period of months to years, depending upon the species. In addition to natural mortalities during ocean residence, Snake River fall chinook salmon are harvested in ocean commercial troll and recreational hook and line fisheries from Alaska to northern California. Current sampling techniques indicate that Snake River spring and summer chinook salmon are taken in ocean fisheries at extremely low rates and that sockeye salmon are rarely taken in ocean fisheries. Historically, a significant harvest of adult fish occurred between the mouth of the Columbia River and the Snake River. Additional human-induced mortalities result from the upstream passage of adults through eight hydroelectric dams between the mouth of the Columbia River and the Snake River basin above Lower Granite Dam. Adults successfully completing the journey back to their natal areas are the spawners for the next generation.

4.2.2 Egg-to-Smolt Stage

Salmon eggs are deposited in excavated nests called redds and covered with a layer of gravel. The eggs incubate in the gravel over winter, with the young salmon hatching and migrating up into the water column in the spring of the subsequent year. The calendar year in which the eggs are deposited is referred to as the “brood year” throughout this report. For salmon, this corresponds with the year the adults return upstream to spawn.

Juvenile salmon spend from several months to a year rearing in fresh water. Near the end of the freshwater rearing period, they begin the process of smoltification, a physiological change that allows them to adapt to seawater. As juvenile salmon begin smoltification, they move downstream from natal areas to begin their migration to the ocean. Survival from egg to migrating juvenile correlates strongly with habitat and climatic conditions. The Snake River tributaries used by listed salmon stocks exhibit a wide range of habitat conditions, from relatively pristine wilderness areas to tributaries drastically altered by human activities such as logging, mining, agricultural practices, and development.

4.2.3 Downstream Migration Stage

Snake River spring/summer chinook salmon and most steelhead migrate to the ocean in the spring of their second year of life. “Migration year” is used to refer to the calendar year during which this movement takes place. The timing of the spring migration occurs during the spring and early summer periods, coinciding with snowmelt in the upper drainages. Migration conditions have been drastically altered by human activities: the development of major upstream storage reservoirs in the Snake and Columbia river basins has changed the shape of the annual hydrograph. Although spring migrants still benefit from the highest annual flows, the flows are much reduced compared to the conditions under which these species evolved. In addition, the major hydroelectric projects have created a series of mainstem reservoirs that are characterized by relatively slow-moving water. Smolts moving though these reaches are subject to predation from resident fishes and birds. In the case of Snake River fall chinook salmon, changes in water temperature associated with various flow regimes and water usage alter migrational timing.

Passage through the dams themselves also results in mortalities. However, a major portion of the Snake River migrants are collected at the upper-most mainstem dams and transported around the hydrosystem, thus avoiding direct losses from passage through multiple dams. Juveniles migrating downstream pass dams via several pathways (turbines, bypass systems with tailrace outfalls, and spillways), each with its own mortality rate. Although the spillway passage route generally is the safest route for passing dams, under conditions with high spill levels, it also poses risks to anadromous fish because it can result in exposure to elevated levels of total dissolved gas. **Section 4.3** discusses in detail how current hydrosystem operations impact salmon, as well as the actions taken to date to mitigate undesirable effects.

4.2.4 Estuarine/Early Ocean Stage

Like salmon runs from other parts of the Columbia River Basin, Snake River salmon are dependent upon conditions in the estuary and the nearshore ocean during the critical first few months of their saltwater life. Relatively little is known about this phase of their life, other than survival rates inferred from tagging studies. Typically, a proportion of the production from a particular brood year ("jacks" and "minijacks") returns to the Columbia River after a few months to one year in seawater. The rate of return of jacks may provide a good indication of the strength of future year classes. Adults return to spawn after two, three, four, or more years at sea and the cycle continues.

4.3 Qualitative Overview of the Likely Effects of Hydrosystem Options on Anadromous Salmonids

In assessing the potential effects of alternative hydrosystem actions on listed Snake River salmonids, NMFS focused primarily on quantitative analyses whereas USFWS, in their Fish and Wildlife Coordination Act Report (FWCAR), focused on the qualitative effects of these actions on all potentially affected species. **Section 4.3** is extracted and condensed from the FWCAR (USFWS 1998). The references cited in the following excerpt from the FWCAR are <u>not</u> listed at the back of this report but are listed in the FWCAR itself.

4.3.1 Effects of Existing Operations on Anadromous Salmonids

> Construction and operation of the lower Snake River dam and reservoir system have affected anadromous salmonids in several ways. These include inundation of spawning habitat, changes in migration rates and conditions of juvenile fish through the reservoirs and at the dams, changes in adult migration conditions, and improved habitat for predators of juvenile salmonids. Hydrosystem effects include both direct (e.g., turbine morality) and indirect effects (e.g. delayed mortality, due to such mechanisms as changes in estuary arrival times)" (Marmorek et al. 1996).
>
> The lower Snake River dams have created reservoirs that affect juvenile salmonid migration by reducing water velocity and disrupting migration timing (Raymond 1979). Higher water temperatures, decreased turbidity, and increased predator populations in reservoirs in combination with the increased passage time through these bodies of water have resulted in greater mortality of juvenile salmonids during downstream passage. Salmonids that are not able to migrate during the critical smolt period may remain in the reservoirs and revert to a freshwater form.

Under the Existing Systems Alternative (alternative A1, **Table 3.4-1**), the lower Snake River dams would remain in place and continue to operate without major change. Some structural changes would be made to improve conditions for migrating adult and juvenile salmonids. Effects on anadromous fish are expected to be similar to those now experienced by all species and runs of fish. Juvenile salmonids would continue to be collected and transported to the lower Columbia River for release or would remain in the river to migrate after passing the dams via spill, collection and bypass to the river, or through the turbines. Fish would be delayed most during low flow periods or when no spill occurred. This is the time when collection and transportation would be most desired. The planned improvements to the existing dam system, such as extended length screens and spill deflectors, which result in greater fish passage efficiency, may increase the survival of juvenile fish. However, juvenile fish would continue to be affected by reservoir conditions that delay migration, increase predation, and subject them to adverse water temperatures unless adequate flows are provided during the migration season. Juvenile salmonids would continue to be impacted by the extremely high levels of total dissolved gas when uncontrolled spill conditions exceed the gas reducing capability of the spill deflectors.

Conditions at the main ladder entrances would continue to be improved to enhance the upstream passage of adult salmonids. For example, the auxiliary water supply at Ice Harbor and Lower Granite dams would be improved to attract adult salmonids to the fishway entrances. Passage delay would occur at each dam, especially during high flow years when fish have difficulty finding fish ladder entrances. High flows and uncontrolled spills would likely cause adult fish to fall back over the dams after exiting the ladders and would also subject them to high levels of total dissolved gas during periods of involuntary spill. Adult salmonids have also exhibited “head burns,” a condition in which the head appears raw or blistered and which may cause increased mortality before spawning. The cause of head burns is unknown, but is suspected to be dam related. This situation would continue to occur until its cause is determined and removed.

The reservoirs in the lower Snake River have inundated about 140 miles (225 km) of free flowing river which formerly provided spawning habitat for fall chinook salmon. Limited fall chinook spawning now occurs in the tailraces of Lower Granite and Little Goose dams near the juvenile fish bypass outfalls. Spawning may also occur in the tailrace of Lower Monumental Dam. Under the Existing Systems Alternative the lower Snake River reservoirs would remain in place and former spawning habitat would continue to be inundated. The limited fall chinook salmon spawning that now occurs in the dam tailraces would continue.

4.3.1.1 Effects on Migration

Development of the dam and reservoir system in the Snake River Basin has changed the hydrograph and migration corridor in the lower Snake River. Construction of the lower Snake River dams has changed this stream from a free-flowing system into a series of slack water reservoirs. In addition, storage reservoirs in the upper Snake River Basin have reduced peak flows that occur in the spring and early summer. Reservoirs now capture and store water for irrigation or later release for power generation during fall and winter months.

The reservoirs have also altered the cross sectional area of the migration corridor in the lower Snake River. The lower Snake River from Ice Harbor Dam to the head of Lower

Granite Reservoir has been changed from its former narrow channel to wider and deeper reservoirs. This has slowed water velocity in this reach of river and increased the time required for water to travel through the reservoirs compared to a free-flowing system. Lowered water velocities in the reservoirs have slowed migration rate of juvenile salmonids through the Snake River (Sims and Ossiander 1981).

Migrating juvenile salmon and steelhead smolts rely primarily on passive transport by water currents and generally do not actively swim downstream. Their successful downstream migration depends on river flow and water velocity which determine how fast they move through the Snake and Columbia rivers to the estuary. Anadromous salmonid smolts are physiologically able to make the transition from freshwater to saltwater during a limited period of time and must reach the estuary within this period. Delays in the downstream migration of juvenile salmonids may affect their ability to successfully make this transition. Delays in migration can subject smolts to higher temperatures. In addition, delayed migration through slow-moving water in reservoirs likely leads to increased predation by fish such as pikeminnow and smallmouth bass, which become more active at higher water temperatures.

The anadromous fish of the Snake River System historically used the increasing flows of the spring and early summer freshet and migrated seaward during this period. Mains and Smith (1964) found that the major period of downstream migration by chinook salmon occurred in the spring and corresponded to the spring freshet during sampling in 1954 and 1955. They noted that downstream movements of chinook fingerlings appeared to be influenced by increases in flow. They also indicated that while temperatures may have played a role in starting the downstream migration of chinook salmon, the first spring freshet was the main factor responsible for stimulating downstream migration. The Snake River discharge required to begin chinook salmon migration was about 70,000 cubic feet per second (cfs) in both years.

During the period before the Snake River dams were built, smolts took 22 days to travel from the Salmon River in Idaho to the Columbia River downstream of Bonneville Dam (Ebel 1977). Since construction of the Snake River dams, the migration time from the Salmon River to the lower Columbia River has been increased to as much as 50 days (Ebel 1977).

The travel time and survival of spring/summer chinook salmon smolts through the Snake River has been found to be related to river flow and water velocity. Sims and Ossiander (1981) estimated smolt travel time and survival for the period 1973 to 1978. They found that travel times for chinook salmon and steelhead were related to river flow and that faster migrations occurred during years of higher river flows. They also found a positive correlation between the average smolt survival at the each of the Snake River dams and flows at Ice Harbor Dam during the period of peak migration.

Raymond (1979) also found that the survival of Snake River smolts was much lower in years when flows and spills were low than in years of higher flows and spills. Raymond (1988) also examined the survival rates of return of adult chinook and steelhead to the Snake River and concluded that the juvenile to adult survival rates of fish that had migrated out during the years 1962 to 1964 had declined because of hydroelectric development.

Petrosky (1991) examined smolt to adult survival rates for Rapid River hatchery and Marsh Creek wild populations of Snake River spring chinook from 1977 to 1987 and compared them with flows at Lower Granite Dam. Petrosky found a positive relationship between migration flows during smolt outmigration and the return rates of these fish which tended to substantiate the flow and smolt survival relationship of Sims and Ossiander (1981).

Delay also occurs when juvenile salmonids approach and pass dams. Migration rates of subyearling fall chinook salmon through the lower Snake River reservoirs appear to decrease as they approach the dams. Venditti et al. (1998) found that the median migration rates of radiotagged juvenile fall chinook salmon decreased from more than 12.4 miles (20 km) per day in upper Little Goose Reservoir to between 6.2 and 9.3 miles (10 and 15 km) per day in the middle reservoir to about 0.6 mile (1 km) per day in the lower reservoir or forebay during studies conducted in 1995, 1996, and 1997. They also found that 10 to 20% of these fish spent more than a week in the forebay. The decreased migration rate was attributed to declining water velocities the fish encountered as they approached the dam. The delayed fish displayed two patterns of movement: one involved repeated crossings of the forebay; the other involved fish moving back upstream as far as nine miles (14 km) after first entering the forebay. The additional delay in migration displayed by up to 20% of the subyearling fall chinook could subject them to additional predation losses and high water temperatures at each dam that these fish pass.

The lower Snake River dams and reservoirs have changed migration conditions for adult salmon and steelhead. Anadromous fish now face an altered system that involves entry into fishways, passage through fish ladders and reservoirs, and altered flow and temperature regimes. The cumulative loss of adult salmonids as they pass the eight dams and reservoirs in the lower Columbia and Snake rivers can be significant. Adult fish losses can be caused by delayed migration, fallback through turbines, and delayed mortality caused by marine mammal predation injuries, gillnet interactions, and disease (NMFS 1995). Based on an analysis of radiotagging studies, the NMFS has estimated that about 39.3% of the adult fall chinook, 20.9% of the spring/summer chinook, and 15.4% of the sockeye are lost during passage through the eight dam and reservoir projects in the lower Columbia and Snake rivers (NMFS 1995).

Adult passage can be delayed at the Snake River dams. The average delay for spring/summer chinook salmon at each lower Snake River project was found to be one to three days when no spill was occurring and five to seven days during high spill (Turner et al. 1983, 1984 in NMFS 1995). During 1993, the median delay was from 0.6 to 1.2 days during periods of no spill to spill of 40 to 80 thousand cubic feet per second (kcfs) (Bjornn et al. 1994 in NMFS 1995).

The total passage times through the lower Snake River for adult spring and summer chinook salmon are not believed to have increased since the construction of the four Corps of Engineers dams. Bjornn et al. (1998) reported that the overall time for radiotagged spring/summer chinook salmon to migrate through the lower Snake River (about 6.4 days) was comparable to that of pre-dam conditions. Upstream migrants were slowed at dams, but migrated through reservoirs at a faster rate than through free-flowing rivers.

Adult salmonids that pass the lower Snake River dams may fall back over the spillways, through the turbines, down the fishways, or through the navigation locks. Fallback has been documented in studies at all of the lower Snake River dams and has ranged from about 4 to 40% during studies (Bjornn and Peery 1992). Radiotracking studies indicate that salmon that fell back over one or more dams were less likely to complete their migrations to hatcheries or to the spawning grounds than fish that did not fall back (Bjornn et al. 1998). Some of the fish that fall back may also have strayed into the Snake River from other areas such as the Hanford Reach of the Columbia River. Those fish may fall back past the dams to reach their proper spawning areas.

Large volumes of spill during conditions of involuntary spill can delay the upstream migration of adult salmonids by making fish ladder entrances difficult to locate. Adult passage counts at the lower Snake River dams typically is reduced during times when flows are high and uncontrolled spill occurs. However, voluntary spill that is provided to improve juvenile fish passage does not appear to affect upstream migrating adult salmon. Bjornn et al. (1998) reported that nighttime spilling for juvenile fish passage at Ice Harbor and Lower Monumental dams resulted in adult passage rates that were similar to those at Little Goose Dam where spill was not provided. Voluntary spills proposed in the NMFS' 1998 Supplemental Biological Opinion are not expected to adversely affect adult salmonid passage at any of the lower Snake River dams based on preliminary information from radiotracking studies conducted by University of Idaho staff (NMFS 1998).

A limited number of fall chinook salmon presently spawn in the tailraces of some lower Snake River dams. Salmon embryos were observed when a site was dredged at Lower Monumental Dam in 1992 (Dauble et al. 1994). However, salmon redds have not been found during surveys that were made in following years. Fall chinook salmon redds have been observed downstream from Lower Granite and Little Goose dams. Spawning is presently occurs in limited areas near the juvenile fish bypass outfalls at both dams (Dauble et al. 1995). Fall chinook salmon would continue to spawn downstream from Lower Granite and Little Goose dams and possibly Lower Monumental Dam in the future with continued operation of the lower Snake River dams.

Concerns have also been raised about the affect high water temperatures in fish ladders has on the migration and survival of adult salmonids. Water temperatures are often higher than 68°F (20°C) and have the potential to delay adult fish migration or to increase the mortalities of adult chinook salmon and steelhead (Bjornn et al. 1997). Water temperatures in the fish ladders have been monitored in the fishways and forebays of Lower Granite and Little Goose dams to see if there is a need to control temperatures. Fish will be examined to determine the relationship between high water temperatures and fish passage at dams.

4.3.1.2 Current System Operations that Mitigate Undesirable Hydrosystem Effects

Two types of spill, involuntary and voluntary, can take place at the lower Snake River dams. Involuntary spill occurs when high river flow exceeds the hydraulic capacity of the dam's turbines or when a lack of electrical power demand reduces the volume of water passing through the turbines. Voluntary spill is a controlled operation that can be started or stopped at any time. Voluntary spill is the type of spill that is provided for juvenile fish passage.

The NMFS' 1995 FCRPS Biological Opinion specifies that water be spilled at the lower Snake River dams to increase the fish passage efficiency and survival of juvenile salmonids when they pass the dams. Fish passage efficiency is the percentage of juvenile salmonid migrants that pass a dam by routes other than turbines. These routes can include spillways, mechanical collection and bypass systems, fish ladders, ice and trash sluiceways, and navigation locks.

Previous studies at Columbia and Snake river dams have shown that juvenile salmonids survive spillway passage at a higher rate than passage through turbines. Studies have shown that mortalities of juvenile salmonids passing through turbines have ranged between 8 and 32% while the mortality rate of fish passing via spillways was between 0 and 4% (CBFWA 1995). The mechanical collection systems at the lower Snake River dams are not able to divert all of the migrating juvenile salmonids away from turbine intakes and into fish bypass systems. Passage through the mechanical collection and bypass system can also result in injury and mortality to juvenile migrant salmonids. However, the survival benefits of transportation are assumed to outweigh the negative effects of collection and transportation. Additional mortality due to increased predation by birds or fish can also occur at bypass outfalls. Fisheries agencies have, therefore, recommended spilling to improve the overall survival of juvenile salmonids passing dams at mainstem Columbia River dams and the lower Snake River dams. Controlled spill programs have been in effect since 1983 at the mid-Columbia River dams operated by the Chelan, Grant, and Douglas county public utility districts and since 1989 at some Corps' dams (CBFWA 1995).

The NMFS' 1998 Supplemental FCRPS Biological Opinion specifies that spill is to occur at the lower Snake River dams during the spring and summer juvenile fish migration seasons. The planning dates for spill are April 3 to June 20 for the spring season and June 21 through August 31 for the summer season. Voluntary spill would occur when the seasonal average forecasted flows at Lower Granite Dam are projected to exceed 85 kcfs during the spring migration period. During the juvenile fall chinook salmon migration period (June 21 to August 31) spilling is recommended only at Ice Harbor Dam.

Spilling of water can cause supersaturation or increased levels of total dissolved gas (TDG) downstream from the dams. High TDG levels can create bubbles in the bodies of fish and other aquatic organisms when the gases come out of solution. This may cause injury or death to fish and other aquatic life at high levels. State of Washington water quality standards limit TDG levels in the Snake River to 110%. However, the Washington Department of Ecology (WDOE) issued waivers of the standard in 1995, 1996, 1997, and 1998 to the NMFS to allow TDG levels of up to 120% in the tailraces and 115% in the forebays of the lower Snake River projects. Biological and physical monitoring has been required as a condition of the waivers. In addition, the WDOE required that a total dissolved gas management plan be developed as a condition for issuance of the TDG waiver in 1998.

In general, studies and biological monitoring that have been conducted show that juvenile and adult salmonids and resident fish are not adversely affected by total dissolved gas levels below 120%. Biological monitoring has documented the effects of exposure to varying levels of TDG upon juvenile and adult salmonids and other fish. In 1995, when the 120% TDG limit was exceeded very few times and 130% TDG was exceeded only at Ice Harbor Dam, the incidence of gas bubble trauma (GBT) in juvenile fish was very low and none exhibited severe signs of GBT. In 1996, the 120% TDG level was exceeded 357 times and the 130% level was exceeded 113 times primarily as a result of involuntary spill caused by high runoff. Severe signs of GBT were observed in 0.12% of the juvenile fish examined. Extremely high flows in the Columbia River in 1997 again resulted in involuntary spill and extended periods of high TDG levels greater than 120 and 130%. In 1997, TDG levels exceeding 120% were recorded 350 times and levels higher than 130% were recorded 162 times. Severe signs of GBT were seen in 0.27% of the juvenile salmonids that were examined (NMFS 1998).

About 0.1% of 6,312 adult chinook salmon that were examined at Lower Granite Dam in 1997 exhibited signs of GBT. However, sockeye salmon and steelhead that were examined at Bonneville Dam showed higher incidences of GBT than chinook salmon at that site. Highest incidences of GBT were noted during the first half of June when involuntary spill and flow were at their maximums (NMFS 1998). Data from the monitoring stations at Skamania, Washington and Warrendale, Oregon indicate that average TDG levels remained higher than 129% throughout the first half of June, 1997. The minimum TDG reading at these stations during this time was more than 127%.

Based on the results of the biological monitoring program it appears that the TDG levels of 120% in the tailraces and 115% in the forebays as authorized in the waivers issued by the WDOE do not threaten the survival of migrating salmonids. The NMFS has recommended continuation of a spill program with modifications in its 1998 Supplemental FCRPS Biological Opinion. The NMFS has recommended that spill should be maximized to the gas cap limits at the lower Snake River projects with the actual dates of spill to be determined each year by the Technical Management Team (TMT) based upon in-season monitoring information. Spill volumes and hours of spill at each of the lower Snake River projects are shown in Table 4.3.2-1.

Under the Existing Systems Alternative, the lower Snake River dams would remain in place and the voluntary spill program would continue. Involuntary spill would occur whenever river flow exceeded dam powerhouse capacity or when there was a lack of a market for electrical power. Spill deflectors (flip lips) would be installed at all the dams and would help maintain lower total dissolved gas levels at higher volumes of spill. However, total dissolved gas levels would not be controllable when river discharge exceeded the total capacity of the spill deflectors and powerhouses until the Corps' Dissolved Gas Abatement Program was implemented. Under those conditions, total dissolved gas concentrations would exceed state water quality standards of 110%or the waiver levels of 120%. Juvenile fish mortality would increase at higher levels of spill and total dissolved gas, especially when total dissolved gas levels were greater than 130%.

Involuntary spill usually occurs during the spring and early summer runoff and would have the greatest affect on spring migrants. These would include juvenile steelhead, spring and summer chinook, sockeye, and coho. Summer migrants, such as subyearling fall chinook, would be less affected by high flows and involuntary spill.

Adult steelhead which overwinter in the Snake River, and spring and summer chinook and sockeye salmon could be present when spill occurs. Based on adult fish monitoring results to date, chinook salmon would not be greatly affected by high total dissolved gas levels. Sockeye salmon and steelhead would be more affected than chinook when total dissolved gas levels were high based on the results of sampling conducted at Bonneville Dam in 1997.

Long-term measures to reduce TDG levels are being investigated by the Corps through its Dissolved Gas Abatement Study. The Corps has identified several alternative measures that have the potential to reduce total dissolved gas. These include spillway deflectors with raised tailraces, raised stilling basins with raised tailraces, raised stilling basins, spillway deflectors, raised tailraces, submerged spill discharges, submerged discharges with deflected spill, raised stilling basins with deflectors, additional spillway bays, side channels, and raised stilling basins with raised tailraces and deflectors (Corps 1997; Northwest Hydraulic Consultants 1998).

The Dissolved Gas Abatement Study is presently scheduled for completion in September of 1999 or 2001 depending on the extent of biological research conducted. Implementation of any of the feasible gas abatement alternatives would require additional study and development of final designs. This process would require an additional three years before the implementation process could begin.

Reduced flow velocity through the reservoirs is believed to have contributed to the decline of Snake River salmon (NMFS 1995). Slow passage of water through the reservoirs prolongs the migration time of juvenile salmonids through the lower Snake River System. This increases their time of exposure to predators and to higher temperatures. Higher temperatures can increase predation rates upon juvenile salmonids and make them more susceptible to disease.

The effect of streamflow in the Snake River on the rate of juvenile salmonid migration and survival has been examined by several investigators. Increased streamflows can reduce the travel time of steelhead smolts and both yearling and subyearling chinook salmon (Berggren and Filardo 1993). Giorgi et al. (1997) reviewed flow augmentation for the years 1991 to 1995 and found that flow augmentation during those years could substantially decrease water particle travel time in Lower Granite Reservoir in the summer when natural runoff is low.

Beuttner and Brimmer (1996) found a significant relationship between migration rates of juvenile chinook salmon and steelhead and increases in river discharge in the Snake River. Detections of PIT-tagged fish showed increased migration rates for hatchery and wild chinook salmon and hatchery and wild steelhead from the Snake and Salmon rivers to Lower Granite Dam with increases in river discharge. Their results for 1995 studies showed that a two-fold increase in river discharge from 50,000 to 100,000 cfs resulted in a 12 fold increase in migration rate of hatchery chinook salmon through Lower Granite Reservoir. Wild chinook migration rates were increased by 4.6; hatchery steelhead by 2.1; and wild steelhead by 2.4 times.

Smith et al. (1997) found a strong and consistent relationship between flow levels and travel times for chinook salmon and steelhead where higher flows were associated with shorter travel times. They also noted an increase in the survival of yearling chinook salmon and steelhead in study reaches upstream from Lower Monumental Dam were higher in 1996 than in previous years and attributed this in part to higher flows. Increased flow in the Snake River is also likely to reduce predation on juvenile salmonids during their outmigration through the reservoirs. Bennett et al. (1996) reported that substantial variation in predation on subyearling chinook salmon in Lower Granite Reservoir occurred from year to year. Higher predation was noted during low flow years when water temperatures were higher and water clarity was greater. They noted the importance of flow augmentation during low flow years to maintain higher flows through Lower Granite Reservoir during June and July. An analysis of PIT tag interrogations showed higher numbers of tagged fish were detected at Lower Granite Dam during high flow years. They indicated that possible benefits of increased flow included lower predation rates and improved migration through Lower Granite Reservoir.

Flow augmentation is provided under requirements of the NMFS' 1995 and 1998 Supplemental FCRPS Biological Opinions. This is accomplished by drafting Dworshak Reservoir down to an elevation of 1,520 feet through August 31, by the U.S. Bureau of Reclamation (BoR) providing 427 thousand acre-feet of flow augmentation from the upper Snake River, and by having the Idaho Power Company provide stored water from

Brownlee Reservoir to help meet flow objectives at Lower Granite Dam. These actions are presently the main measures used to augment flows in the lower Snake River.

Flow objectives at Lower Granite Dam have been established on a sliding scale for the spring and summer. The spring flow objective ranges between 85 and 100 kcfs on a linear sliding scale when the April to July water volume runoff forecast for the Snake River at Lower Granite Dam is between 16 and 28 million acre feet (MAF). The flow objective is for at least 100 kcfs when the volume runoff forecast is more than 28 MAF.

The summer flow objective is determined similarly. When the April to July volume runoff forecast is between 16 and 28 MAF, then the summer flow target is between 50 and 55 kcfs. When the forecast is greater than 28 MAF, then the summer flow target is at least 55 kcfs.

The TMT, which is composed of representatives of fishery management agencies and the Federal dam management and power marketing agencies, addresses flow augmentation requirements during the fish migration season. The TMT makes weekly flow recommendations for the Columbia and Snake River dam and reservoir system based on considerations for the following: (1) the timing and number of fish migrating; (2) the probability that enough water will be available to augment flows for juvenile and adult fish throughout the migration season, or that in low water years available water will be allocated according to designated priorities among different species and life stages; and (3) instream water temperatures and the effect augmentation would have on future temperature conditions and fish resources.

The Snake River System reservoirs and streamflows have been managed for three years under the NMFS' 1995 Biological Opinion's requirements for fish passage. In 1995, the seasonal flow targets were met for both the spring and summer. Daily flows were lower than the spring target during April and early May and lower than the summer target after mid-July. The years 1996 and 1997 were high runoff years in the Snake River Basin, which should have provided ample supplies of water for flow augmentation. In 1996, the Snake River storage reservoirs were operated to meet flood control requirements during the spring and early summer and spring flows were the result of flood control operations. The spring flow target at Lower Granite Dam was exceeded. The summer flow target was met on a seasonal basis, but flows varied during this period and were below the target at times (FPC 1997).

Under the Existing Systems Alternative, flow augmentation would likely continue according to the NMFS' 1995 and 1998 Supplemental FCRPS Biological Opinions. Flow augmentation would continue to be provided as specified each week by the TMT. The ability to meet flow targets would be subject to hydroelectric, flood control, irrigation, navigation, and recreational needs. Under these conditions, it is not likely that flow targets for fish passage could be consistently met on a weekly or daily basis.

Presently, most of the water available in Dworshak and Brownlee reservoirs has been directed toward flow augmentation for juvenile fall chinook migration. The NMFS has indicated that flows for juvenile summer migrating salmonids would be given priority over flows for adult fall chinook salmon (NMFS 1995). This priority is not likely to change unless additional water becomes available or the summer travel time of migrating subyearling fall chinook is minimized.

Flow augmentation would continue to increase subyearling fall chinook salmon survival during seaward migration. Summer flow augmentation, especially from Dworshak Reservoir, increased subyearling chinook salmon survival during 1992 to 1995 by limiting thermally induced mortality in dry years and reducing predation under all flow conditions (Connor et al., in preparation). However, releases of cold water from Dworshak Reservoir for summer flow augmentation may affect growth and time of migration of fall chinook salmon in the Clearwater River. In 1994, flow augmentation included 26 days of releases of 47°F (8.7°C) water that dropped water temperature in near shore areas of the Clearwater River from about 61°F (16°C) to 54°F (12°C) when parr were present (Connor et al. 1997). Such a drop could possibly affect smoltification (Arnsberg and Statler 1996 and Connor et al. 1996) and may have led to the high proportion of fish that emigrated as yearlings from 1994 Clearwater River releases of fish (W.P. Connor, USFWS, unpublished data). The problem that occurred in 1994 was corrected by the TMT in 1995 by shaping flow augmentation to benefit both Snake and Clearwater river fish (Connor et al. 1997). In 1996, however, another potential problem was recognized. A decrease in overall survival for subyearling fall chinook salmon in the Snake River was noted when compared to 1995 (Connor et al. in press). One possible cause of this decrease was a reliance on early releases of 68°F (20°C) water from Hells Canyon complex to meet the flow target at Lower Granite Dam. Flow augmentation is a complex process that is difficult to implement.

Transportation has been used to try to reduce the losses of juvenile Snake River salmonids during their downstream migration since 1965 when experiments were started on chinook salmon and steelhead (Ebel 1974). The NMFS conducted research from 1968 to 1989 to determine the comparative survival of juvenile chinook salmon and steelhead that were transported from Snake River dams with those that migrated in the river (Ward et al. 1997). Based on its interpretation of the results of these studies, and with the concurrence of all the fishery management agencies, the NMFS began to transport all fish that were collected at Lower Granite Dam in 1976 and at Little Goose Dam in 1977 (Park 1985 in Ward et al. 1997). Mass transportation of juvenile salmonids was implemented as an operational program by the Corps in 1981 (Independent Scientific Group 1996). In the Snake River, fish are presently collected and transported at Lower Granite, Little Goose, and Lower Monumental dams.

The NMFS' 1995 FCRPS Biological Opinion requires that chinook salmon smolts collected at Lower Granite, Little Goose, and Lower Monumental dams be transported except when transportation operations do not meet criteria established in the Corps' Juvenile Fish Transportation Plan. Presently, most juvenile salmonids that are collected at Lower Granite, Little Goose, and Lower Monumental dams are transported. In 1996, about 11,222,600 fish (85.8% of all fish collected) were transported by barges, trucks, or small tanks loaded onto pickup trucks (Corps 1997). The fish transport program begins the last week in March at Lower Granite Dam and starts at Little Goose and Lower Monumental dams based on collection numbers at Lower Granite Dam and expected migration times to the lower two dams (Corps 1998). Fish transport normally ends on October 31 at Lower Granite, Little Goose, and Lower Monumental dams.

According to the Fish Passage Plan, barges are the main mode of fish transportation during the peak migration season, which begins when total collection at a dam reaches 20,000 fish per day. Truck transportation is used before and after the peak migration period. After collection, fish may be held in raceways for up to two days for transport. Collected fish are transported daily during the peak passage period. When the numbers offish decline, during the late summer, when less than 500 fish are collected daily, fish are loaded into sample holding tanks at Lower Granite and Little Goose dams, examined for the Smolt Monitoring Program, and loaded into trucks. At Lower Monumental Dam, fish are routed to sample tanks when their numbers become low enough for handling. Fish are loaded from raceways to large trucks or transferred from examination facilities to mini-tanks for transportation. During the summer period, collected fish may be sampled every other day to reduce stress.

Fish are released from barges at night at selected sites downstream from Bonneville Dam between the Skamania light buoy at about river mile 140 to Warrendale, which is at about river mile 141. During spring, trucked fish are released at Bradford Island next to the Bonneville Dam first powerhouse. From mid-June to the end of the transportation season, trucks and mini-tanks are to be loaded onto barges downstream from Bonneville Dam and transported to a mid-river site where transported fish would be released.

There has been disagreement over the reliance on transportation as the main strategy for recovering Snake River salmonids. While the NMFS has supported transportation of Snake River fish, the state and tribal fishery managers have favored an approach that relies less on transportation as the main tool and that uses spill to increase juvenile fish survival. In response to these concerns, the NMFS included a provision in the 1995 Biological Opinion that spill be initiated when the average flow in the Snake River reaches 85 kcfs at Little Goose and Lower Monumental dams and 100 kcfs at Lower Granite Dam. The NMFS' 1998 Supplemental FCRPS Biological Opinion further addresses the spill and transportation issue by taking a "spread the risk" approach in which both transportation and spill are provided for migrating juvenile salmonids. It has reduced the spill trigger for steelhead at Lower Granite Dam to 85 kcfs.

Concerns about the fish transportation program have included: (1) delayed effects on juvenile fish after they are released from barges or trucks including stress; (2) disease; (3) susceptibility to predation; (4) impaired homing; (5) delayed mortality; (6) whether the timing of downstream migrants would be disrupted by transportation; and (7) whether the smolt to adult return rate shown by transported fish was sufficient to provide recovery of Snake River salmon.

The NMFS has summarized the results of past juvenile fish transportation studies in its 1998 Supplemental Biological Opinion. Some of their findings include: (1) Transportation helps reduce the number of juvenile salmonids killed in the existing hydropower system and increases the number of returning adult fish; (2) Straying responses of transported fish are small and no greater than natural rates; (3) There are no conclusive research results showing that transportation improves returns to spawning grounds or provides sufficient adult return rates to recover upriver runs; (4) No precise data have been collected on juvenile mortality during or following transportation; and (5) There does not appear to be large scale predation on smolts immediately after their release from barges.

In 1997, the Independent Scientific Advisory Board (ISAB) was asked three questions related to the transportation of juvenile salmonids from the Snake River (ISAB 1998). The questions asked were: (1) if there were significant differences in the survival to adult returns of salmon and steelhead that were transported compared to those left in the river; (2) if there were significant differences in the straying rate of fish that were transported as juveniles compared to those left to migrate in the river; and (3) what the likelihood that collection and transportation of salmon and steelhead at the lower Snake River projects and McNary Dam in 1998 will result in an increased return of adult fish compared to those left to migrate in the river. In response to the question of survival to adults, the ISAB indicated that transportation would probably improve the survival of some stocks of anadromous fish. However, the ISAB qualified this response by noting that it was not known which stocks or populations would benefit and which would suffer from transportation. The ISAB indicated that differences in the straying rates between transported juvenile fish and those left to migrate in the river had been observed. Higher rates of straying of transported fish appeared to have been related to inadequate imprinting by juvenile fish. This occurred most often when fish were transported by truck. The ISAB also noted that it was not known whether the differences in straying rates were biologically meaningful.

In its response to the question regarding the likelihood that transportation of fish in 1998 would result in higher returns of adult fish, the ISAB stated that the effects of a combined trucking and barging operation were uncertain if all species, life history types, and populations were considered together. The ISAB emphasized that a single action such as transportation needed to consider the variation within and between populations as well as its average benefit.

The ISAB had three major recommendations regarding transportation of juvenile fish. These were: (1) that a "spread the risk" approach be taken which divides the juvenile migrants between barging and natural migration be taken throughout the 1998 juvenile fish migration; (2) that trucks not be used to transport fish; and (3) that management actions for salmon and steelhead be as population specific as possible.

Presently, studies of fish transportation are continuing. Proposed studies for fiscal year 1999 funding include: (1) the ongoing comparison of transported juvenile fish to those which remain in the river; (2) evaluating transportation of fish to the Columbia River Estuary; (3) evaluating the effects of the procedures of collection, transportation, downstream passage, and post-release survival of outmigrating salmonids; (4) evaluating the migration and survival of juvenile salmonids following transportation; and (5) evaluating the influence of transportation on the homing of spring and summer chinook salmon. Most of these studies will not be completed before the 1999 decision date regarding the lower Snake River dams.

Under the Existing Systems Alternative, transportation of juvenile fish would continue under the operations specified in the NMFS' 1995 and 1998 Supplemental FCRPS Biological Opinions. This requirement is based on NMFS study data that indicate that transportation benefits spring/summer chinook and is likely to benefit sockeye and fall chinook salmon. The NMFS' 1995 FCRPS Biological Opinion specified that all fish collected at the lower Snake River projects be transported unless the TMT recommended otherwise based on credible evidence that migration in the river would be beneficial. The NMFS' 1998 Supplemental FCRPS Biological Opinion removes any flexibility for returning fish to the river and now requires that all juvenile salmonids collected be transported. On-going studies related to transportation, especially the survival comparison of transported fish to those that migrate in the river, would continue until enough data are collected to make a decision regarding future transportation.

In addition to augmenting flow, Dworshak Reservoir water has been used to control water temperature in the Snake River. Dworshak Dam is capable of providing cold water to lower the temperature of the Snake River in the summer. Cold water releases from Dworshak Dam can improve water temperatures in the Snake River for subyearling chinook salmon during the summer provided it is released at the most suitable time. During 1995, cooler water was released from Dworshak Dam for 48 days and helped maintain suitable water temperatures for subyearling chinook in both the Snake and Clearwater rivers. An indirect benefit of the 1995 flow augmentation may have been reduced smallmouth bass predation on subyearling chinook salmon. Anglea (1997) found that smallmouth bass consumption of subyearling fall chinook salmon in Lower Granite Reservoir in 1995 was lower than in 1994. He indicated that flow augmentation from May to mid July of 1995 probably decreased temperatures and increased turbidity compared to 1994.

In 1996, Dworshak and Brownlee reservoirs were operated according to a State of Idaho plan that released more water from Brownlee in July and August and delayed the release of water from Dworshak. This differed from the NMFS' 1995 FCRPS Biological Opinion scheduling of reservoir releases although the total volume of water provided was the same. Flows at Lower Granite Dam decreased in July and increased at the end of August. Lower flows and warm water from Brownlee Dam resulted in water temperatures being 1.8°F higher in July and 0.8°F. higher in August compared to 1995. Higher water temperatures likely contributed to increased mortality of subyearling fall chinook. Dworshak Dam began releasing cooler water in mid-August, but this provided less benefit to fall chinook because it was likely that many of those fish had not survived the earlier high temperatures.

Under this alternative, Dworshak Dam would continue to provide cooler water to help maintain more suitable summer temperatures in the lower Snake River reservoirs for salmonids. Subyearling chinook migration would need to be monitored to determine when fish would be best benefited by flow augmentation from either Brownlee or Dworshak reservoirs. Monitoring of water temperatures in the Snake River and Dworshak and Brownlee reservoirs would be necessary so that the most suitable flows and temperatures could be provided to improve subyearling chinook survival.

Flow augmentation and associated temperature control affect the time and rate at which Dworshak Reservoir is drawn down which in turn affects recreational use in the reservoir. The State of Idaho has expressed its interest in maintaining a high water surface elevation in Dworshak Reservoir during the summer for recreational purposes. Brownlee Dam could be used to shape flow augmentation by shifting the time when water was provided. However, it would not be able to help control water temperature in the Snake River unless it is provided with the capability to draw water from greater depths in the reservoir.

Under the Existing Systems Alternative, there would be no major changes in the water temperature regime of the lower Snake River. The periods of upstream migration and spawning by Snake River fall chinook salmon would not be expected to change. Egg incubation, emergence times of fry, and downstream migration would also remain unchanged.

4.3.2 Existing Operations with Surface Bypass/Collection Alternative

The PATH group indicated in its Retrospective Analysis (Marmorek et al. 1996) that there was not enough existing information to determine if surface collectors could substantially increase the proportion of Snake River salmon that are transported. The PATH group indicates that a surface collector at Lower Granite Dam could potentially increase the total number of smolts transported from the Snake River by 6 to 13% if it is as effective as the surface bypass system at Wells Dam. Presently, the available information is insufficient to determine the potential efficiency of a surface collector at Lower Granite Dam (Marmorek et al. 1996). PATH also states that it is unlikely that the surface collection system at Lower Granite Dam will be as efficient as the Wells Dam facility because of differences in dam configurations. There are basic differences between the Wells Dam surface bypass and the Lower Granite Dam SBC. At Wells Dam the spillgates are directly over the turbine intakes while the powerhouse and spillway are separate. The Lower Granite Dam SBC is an attempt to simulate the system at Wells Dam. The Wells Dam system also uses between 6 and 8% of the total river flow compared to 1 to 2% of the river flow used by the Lower Granite prototype.

4.3.2.1 Current Information

A prototype SBC was installed at Lower Granite Dam in 1996 and was tested during the 1996 and 1997 migration seasons. Additional testing and studies were being conducted during 1998 using a modified collector patterned after the Wells Dam bypass. The modified collector is designated as the Simulated Wells Intake (SWI). A Behavioral Guidance Structure (BGS) was also tested for use with the prototype SBC at Lower Granite Dam. This structure is being tested to determine if it will guide migrating juvenile salmonids toward the SBC and prevent them from entering the powerhouse. The BGS is a steel wall, suspended by floats, that would separate the forebay into two sections in front of the powerhouse at Lower Granite Dam. The prototype SBC is a half powerhouse structure. The BGS was intended to negate the need for a full powerhouse structure by guiding fish to entrances at the north side of the powerhouse.

The prototype SBC was studied by researchers using radio-tagged fish and hydroacoustic techniques to determine its effectiveness in attracting and passing juvenile salmonids. Johnson et al. (1997), using hydroacoustic equipment, found that the SBC entrance efficiency was about 70% for fish within 10 feet (3 m) of the entrance during spring 1997 monitoring. However, they expressed concern that 30% of the fish did not enter the SBC. The effectiveness of the SBC was found to be close to four times greater than spill effectiveness in terms of smolts diverted per volume of water. The incremental benefit provided by the SBC over the screen system itself was found to be relatively small (4 to 7%). Use of both the SBC and the existing turbine intake screen system was found to be necessary to achieve the desired level of fish protection. About 37% of the outmigrant fish are estimated to have passed the test area via a non-turbine route without use of the existing screen system. It should be noted that these estimates are for fish approaching the entrances in front of turbine units 4,5 and 6 and are not an estimate for all of the fish approaching the powerhouse. Most of the fish passing the dam during the monitoring period were hatchery steelhead.

Adams et al. (1997) found that 13% of chinook salmon radio-tagged in 1997 passed within 33 feet (10 m) of the SBC and that 51% of those fish (6% of the total) entered and passed through the structure. They also found that 91% of the other fish that approached within 33 feet (10 m) of the SBC passed under it and into the turbine intake screen and bypass system. Results from the 1996 and 1997 studies showed that many of the fish that approached the SBC at depths of less than 30 feet (9 m) went under or around the structure. Most of the fish that approached at depths greater than 30 feet (9 m) went under the SBC.

Researchers also observed different responses in steelhead, spring/summer chinook, and fall chinook. Hatchery steelhead were found to be guided most efficiently. Chinook salmon which migrate deeper in the water column than steelhead, were guided less efficiently by the surface bypass collector than steelhead. Fall chinook salmon which migrate at greater depths than spring chinook, had the lowest fish passage efficiency, overall combined bypass efficiency, and fish guidance efficiency of the three types of salmonids tested. Radio-tagged fish studies in 1997 found that 42% of hatchery steelhead, 35% of wild steelhead, 41% of all chinook, and 24% of fall chinook that came within 98 feet (30 m) of the area in front of the test turbine units entered the SBC (Adams et al. 1998).

Testing in 1998 was modified based on results of the preceding years' tests and a desire to test the BGS. The SWI was attached to the dam face below the SBC to extend the bottom of the SBC by 20 feet (6 m). The SWI is intended to reduce downward flow in front of the SBC for a distance out to 98 feet (30 m). The intent of this modification is to reduce fish entrainment into the turbine flow and improve the opportunity for fish to find the SBC entrances.

The BGS will be placed in the forebay to direct juvenile migrant salmonids away from turbine intakes 1, 2, and 3. It will be a 1,100-foot-long (335-meter-long) steel curtain that will extend from the BGS upstream toward the south shoreline of the forebay. It will be 79 feet (24 m) deep at the downstream end and taper to 56 feet (17 m) deep at the upstream end.

Results of the 1998 testing program for the SBC and BGS may not be available at the time of this draft anadromous fish assessment. However, the Corps has developed a decision path for future testing of the SBC based on performance targets and criteria. If performance of the hybrid SBC-SWI-intake screens system is comparable to that at Wells Dam, then testing of a dewatering system would be pursued in 1999 as well as retesting to validate the 1998 results. Retesting of the SBC-SWI-intake screen system would be conducted in 1999 if performance was similar to or only marginally better than the existing SBC and screen system. No further testing of the hybrid system would be conducted if the SBC efficiency was less than 40% for chinook or if it was less than the existing system for steelhead. Future testing of the stand-alone SBC-SWI system without intake screens would not continue if its efficiency was less than the existing system. The BGS would be retest in 1999 unless fewer than 70% of the fish are diverted and if no obvious problems are identified.

Preliminary results of 1998 hydroacoustic and radio-tagging studies conducted on spring migrating fish appear to indicate that the combined SBC-BGS has directed more fish to the SBC in 1998 than in 1997. Very preliminary estimates are that about 18% of the fish that came within 98 feet (30 m) of the structure entered the SBC in 1998 spring tests. This compares with about 10% of all fish in 1997 tests (N. Adams, U.S. Geological Survey, personal communication). The BGS appears to guide fish either towards or away from the SBC prototype structure. However, some fish passed through the upstream gap between the BGS and shore and others passed under the BGS. Studies of summer migrants were conducted in July 1998. Preliminary results are not expected to be available until September 1998.

It appears likely the Corps will propose that testing of both the SBC and BGS will continue during 1999. Researchers have indicated that they could complete data collection and analysis from studies that would be conducted in the spring and summer of 1999 before the decision regarding the lower Snake River projects is to be made. Studies in 1999 are likely to involve additional changes to the SBC and BGS to change current patterns near the SWI and reduce passage around and under the BGS. It is likely that future actions would include additional modifications of the SBC and BGS with accompanying studies.

Johnson et al. (1997) noted that the Wells Dam surface bypass took 12 years to fully develop and evaluate.

Additional concerns were expressed during the development of the study design for the SBC and BGS regarding the impacts of these structures on predation upon juvenile fish and migration of adult salmonids. Both of these structures may provide greater opportunities for predation by pikeminnow since they would divert and concentrate juvenile salmonids. There were also concerns about increased predation in the tailrace of Lower Granite Dam. Adult fish could also be affected if they encounter the structures and are delayed or fall back past the dam. Predation studies are being conducted to determine if the SBC or BGS are increasing habitat or opportunities for predators. Researchers are also monitoring the movements of radio-tagged adult salmonids after they leave the Lower Granite Dam fish ladder. Results of the 1996 through 1998 studies do not indicate problems with adult passage or predation on juvenile salmonids due to the presence of the prototype SBC and BGS.

Under the Surface Bypass/Collection Alternative juvenile salmonid passage at the Snake River dams would be improved if this concept is successfully developed from a prototype to a full scale operation. To be successful, the SBC would have to increase the fish passage efficiency at each dam to a level comparable to that at Wells Dam. At Lower Granite Dam fish passage efficiency would have to increase by at least 6 to 13% over that which can be achieved by the existing juvenile fish collection and bypass system. Adult fish migration would not be changed from the Existing Systems Alternative as long as attraction flows to the SBC do not direct fish from the ladder exits to the SBC and the BGS, if it is installed, does not delay upstream migration.

Survival rates of juvenile salmonids after passing through the SBC would vary depending on whether they were returned to the spillway, returned via the fish bypass system, or were collected and transported. The SBC system would need to be dewatered to a manageable volume if fish are to be conveyed to transportation facilities without injury. Dewatering of the bypass system so that juvenile fish can be transported must be addressed in detail to determine its feasibility. Survival of juveniles using these routes of passage to the lower Columbia River would be similar to the survival rates presently observed for those routes.

Juvenile migration rates through the Snake River reservoirs and survival in the pools would not change with the installation of surface collection systems because the total volume of river flow would not differ from that under existing conditions. One of the main potential benefits of the SBC would be to reduce the delay of migrating juvenile salmonids in the forebays of the dams. Forebay delay by subyearling fall chinook salmon, which has been observed at Little Goose Dam, would not change unless the zone of influence affected by the surface collection systems extended to the distance from the dam at which the subyearling fall chinook begin their wandering movements. Kofoot et al.(1997) found that the greatest numbers of juvenile salmonids were found in hydroacoustic monitoring transects located closest to the face of Lower Granite Dam in 1996. However, the closest transect was 100 feet (31 m) from the dam. The zone of separation where fish became entrained in the turbine flow or discovered the surface bypass collector's flow extended 33 feet (10 m) to 66 feet (20 m) out from the collector (Johnson et al. 1997).

Other operations under the Existing Systems Alternative would most likely continue with the Surface Bypass/Collector Alternative. In order to achieve high fish passage efficiencies, the SBC would be used with the existing screen and bypass system in operation. It should be noted that the original intent of the surface bypass concept was to provide a collection or bypass route that was more “normative” and less stressful on juvenile fish and not simply to collect more fish. Survival of juvenile salmonids that are collected by the existing system would be similar to that which is now observed.

4.3.3 Natural-River Drawdown (Dam Breaching) Alternative

The Natural River Drawdown Alternative would eventually restore the lower Snake River to a riverine condition after accumulated sediments are flushed out of the system and into the Columbia River. Dam breaching and drawdown would have short-term and long-term effects on anadromous salmonids. Short-term effects would occur during the actual dam breaching process and afterward as the reservoirs are drawn down and sediments are flushed out. Short-term effects may be adverse to anadromous salmonids, but can be lessened by properly timing work activities, by providing measures to protect fish during construction, and by implementing fish salvage measures.

In the long-term, dam breaching and drawdown would benefit all of the anadromous salmonids in the Snake River Basin. The major effects on anadromous fish would include restoration of spawning and rearing habitat for fall chinook salmon in the lower Snake River, improved migration conditions for juvenile salmonids and lamprey, and unimpeded upstream migration for adult salmonids, lamprey, and white sturgeon. Another major effect would be a significant reduction in mortality and injuries from fish that are now collected, go through turbines, or are bypassed in some other manner. This would also benefit resident fish that use the river as a migration corridor. This increase in survival has the potential to help rebuild anadromous fish stocks in the Snake River. PATH analyses and **Sections 5** and **6** of this report attempt to quantitatively examine the likely effects of drawdown.

4.3.3.1 Spawning Habitat

Restoration of the lower Snake River to a riverine condition would likely reestablish suitable spawning habitat for fall chinook salmon in the reach of river from the head of Lower Granite Reservoir to Ice Harbor Dam. Before inundation, the lower Snake River included a series of pools, riffles and rapids. Islands and side channels were present at numerous locations. A review of survey maps from 1917 shows that the Snake River was a dynamic alluvial river system that had diverse habitats such as spawning, rearing, and adult staging areas. Fall chinook salmon spawning reportedly occurred in the lower Snake River before construction of the lower river dams (NMFS and USFWS 1972; Fulton 1968).

It is likely that fall chinook salmon would spawn in the lower Snake River after finer sediments that have accumulated in the reservoirs are removed, the former riverbed is re-exposed, and gravels are redistributed or flushed of fine materials. The length of time or river discharge that would be needed to reestablish suitable spawning substrate for fall chinook salmon spawning are presently unknown. As noted previously, Hanrahan (1998) indicated that the Lower Granite reach of the lower Snake River would be flushed of fine sediments within five years. Observations of a test drawdown of Lake Mills, a reservoir on the Elwha River in Washington, showed that erosion of exposed sediments occurred rapidly. Modeling of streamflows based on historic records of high, medium, and low flow years indicated that the Elwha River would reform a river channel in about three years (T. Randall, BoR, personal communication).

Establishment or expansion of spawning populations of fall chinook salmon elsewhere in the Columbia River Basin in recent years indicates that they will use suitable habitat when it becomes available and when adult fish return in adequate numbers. Spawning populations of fall chinook have remained stable, increased, or become established at several locations in the Columbia River Basin. These include populations in the Hanford Reach of the Columbia River, the Columbia River in the Pierce/Ives Island area just downstream from Bonneville Dam, the lower Deschutes River, Hell's Canyon and the Clearwater River.

One example of a population that has maintained itself is the Hanford Reach group of upriver bright fall chinook in the free-flowing section of the Columbia River. The Independent Science Group (ISG 1996) has highlighted the contrast between the spawning populations of fall chinook salmon in the Hanford Reach of the Columbia River and the Snake River downstream from Hells Canyon Dam. The Hanford Reach fall chinook population has increased substantially, while the Snake River population has declined to the point where listing under the Endangered Species Act was warranted. Spawning escapement of adult fall chinook to the Hanford Reach has ranged from about 14,000 in 1981 to 105,000 in 1986 and averaged about 40,000 during the period from 1964 to 1991 (Langness et al. 1998). The Snake River fall chinook population has been reduced to the point that available spawning habitat exceeds the number of returning adult fish available to utilize it (Connor et al. 1994).

The major difference in conditions encountered by Hanford Reach and Snake River fall chinook appears to be upstream from the confluence of the Snake and Columbia rivers since both populations pass through the lower Columbia River and its four mainstem dams. The most obvious difference between the Hanford Reach and the lower Snake River is the presence of the four dams and reservoirs. There are several potential causes for the difference in success between Hanford Reach and Snake River fall chinook salmon that were related to the presence or absence of dams and reservoirs. These include differences in survival during juvenile migration, differences in rearing habitat, predation, delay in juvenile migration through the Snake River dam and reservoir system, and water temperature.

Hanford Reach and Snake River fall chinook emerge and begin their downstream migrations at about the same time (W. Connor, USFWS, fishery biologist, personal communication). Delay in passing through the lower Snake River reservoirs appears to be a primary difference between the two stocks of fish. Hanford Reach subyearling fall chinook now reach McNary Reservoir at about the same time Snake River fish arrive at Lower Granite Reservoir. Delayed passage through the Snake River reservoirs results in Snake River migrants arriving at McNary Reservoir later than Hanford Reach fish. In addition, loss to predation also occurs in the Snake River pools.

The Independent Scientific Group discussed differences in suitable habitats for rearing and migration for subyearling fall chinook salmon were as a possible cause for the difference in success of the two populations. Hanford Reach habitat consists of flooded cobble shallows and riparian vegetation which provide abundant insect life while Snake River reservoirs consist of eroding soil banks and riprap which are poor habitats for producing insect food items for subyearling chinook.

Another population of upriver bright fall chinook salmon has become established within the past decade in the Pierce/Ives Island reach of the Lower Columbia River downstream from Bonneville Dam. This population has established itself in suitable habitat evidently from fish that originated from hatchery programs. WDFW has conducted a genetic analysis of fish from this population and determined that they are most closely related to upper Columbia River bright stocks (Marshall 1998). Spawning has occurred in substrates and at velocities similar to those where spawning has been observed at other locations in the Columbia and Snake rivers.

Fall chinook spawning has also been reestablished in the lower Clearwater River and has been increasing in recent years (B. Arnsberg, Nez Perce Tribe, personal communication). Fall chinook salmon now spawn in the lower 45 miles (72 km) of the lower Clearwater River (Groves and Chandler, in press).

The run of Deschutes River fall chinook salmon is another example of a population that has increased. This population has increased and has averaged about 6,000 fish annually from 1990 to 1996 (Myers et al. 1998). Deschutes River fall chinook salmon are also now considered to be part of the same population as Snake River fall chinook. The NMFS has determined that Deschutes River fall chinook are within the Evolutionarily Significant Unit for Snake River fall chinook salmon (Myers et al. 1998). This determination was based on genetic, life history, and ecological similarities and differences of chinook salmon.

Historically, the Snake River is thought to have produced the greater portion of this population compared to the Deschutes River. The Snake River spawning population is estimated to have been about 72,000 during the 1930s and 1940s while the Deschutes River population was much smaller. The situation has changed to the point where the Deschutes River natural spawning population of about 6,000 fish is far larger than the Snake River natural spawning population of about 500 (Myers et al. 1998)

Snake River fall chinook salmon have been observed spawning in areas with characteristics similar to those used by fall chinook for spawning elsewhere in the Columbia River System. Connor, et al. (1993) found fall chinook salmon spawning in areas with physical characteristics typical of spawning sites used by fall chinook in other reaches of the Columbia River. Snake River fall chinook have been reported to use substrates ranging from one to six inches (2.5 to 15.2 cm) in diameter for spawning. They have also been observed spawning at depths ranging from 1.3 to 21 feet (0.4 to 6.5 m) deep. The preferred current velocity for spawning has been found to range between 1.3 to 6.4 feet (0.4 to 2.0 m) per second (Connor et al. 1994; Groves and Chandler, in press).

Snake River fall chinook typically spawn from late October to mid-December as water temperatures decline. Most spawning occurs from late October through November. In recent years spawning in the Hells Canyon reach of the Snake River has started as temperatures dropped to about 57^{o}F (14^{o}C) and ended as temperatures reached about 41^{o}F (5^{o}C) (A. Garcia, USFWS, fishery biologist, personal communication).

It is likely that fall chinook spawning would occur in the lower Snake River after drawdown and riverine conditions are restored provided: (1) suitable substrates become available; (2) suitable temperatures occur during the known spawning time of Snake River fall chinook; and (3) adequate streamflows are available to provide sufficient current velocities. Records from surveys taken before the lower Snake River dams were built will be examined to determine the total area of the potential fall chinook salmon spawning gravels. Flow velocities, modeled by the Pacific Northwest National Laboratory (Hanrahan et al. 1998) and combined with the GIS survey information, will be used to estimate the total maximum potential fall chinook spawning area that could exist after drawdown of the lower Snake River reservoirs.

Monitoring of potential fall chinook salmon spawning sites would be necessary to determine the suitability of gravels and cobbles, water velocities over these sites at known river flows, water flow through potential spawning gravels, and water temperatures during the time of expected fall chinook spawning. Control of river flows and temperatures by releases of water from Dworshak and/or Brownlee dams could be used to provide suitable conditions for fall chinook spawning and incubation.

Fall chinook that originate from upstream locations in the Snake River Basin may naturally establish the spawning population in the restored river reach or fish from other sources, such as the Hanford Reach or Priest Rapids Hatchery, may colonize it. Establishment of the spawning population from Snake River fall chinook stock is essential to maintain the genetic integrity of this population. Supplementation of fish from Lyons Ferry Hatchery, which are included in the Evolutionarily Significant Unit for the Snake River, may be possible to initially establish a spawning population in the lower Snake River.

The Natural River Drawdown Alternative would not change the time of spawning by fall chinook salmon unless major temperature changes occurred. It would remove the heat storage capacity of the four lower Snake Reservoirs and allow faster cooling of the Snake River in the fall. Drawdown could improve migration conditions for adult fall chinook by lowering river temperatures in the summer if combined with water temperature control. In low water years, warm water temperatures in the late summer may stress adult fish in the Snake River. Releases of cool water from Dworshak Reservoir are used to lower water temperatures, but this effect is dissipated by warm water in the reservoirs. Restoration of a riverine condition could provide a greater and faster cooling effect from Dworshak Reservoir releases.

Optimum flows for fall chinook salmon spawning would need to be determined after suitable spawning substrate becomes established since the locations of spawning sites cannot be predicted. Instream flow studies should be conducted to determine the optimum flow conditions for fall chinook salmon spawning, incubation, rearing. Without such information, we would recommend that an interim flow that is near the flow that existed before construction of the Snake River dam and reservoir system be provided. Determination of this flow may be difficult because of the extensive water development project system in the Snake River basin. Flows in the lower Snake River during the fall chinook spawning period are also controlled by water that is released from the Hells Canyon hydroelectric complex and from Dworshak Dam.

4.3.3.2 Juvenile Fish Migration

Juvenile salmonid migration through the lower Snake River is presently much slower than it was under free-flowing river conditions before the dams were built. Breaching of the dams would restore the Snake River to a free flowing condition and conditions for migrating juvenile salmonids would be similar to the pre-dam situation. The Natural River Drawdown Alternative would result in a return to riverine conditions in the lower Snake River. One of the major results of restoring the lower Snake River to riverine conditions would be faster travel by juvenile salmonids though this reach of river to the Columbia River. Faster migration rates would occur with or without flow augmentation. Downstream migration rates would be similar to those for fish in the Hells Canyon Dam to the head of Lower Granite Reservoir.

Passage delays in the reservoirs and at the four lower Snake River dams would be eliminated by breaching the dams. Direct observations of radio-tagged wild and hatchery juvenile steelhead showed that their migration rates in the free-flowing reach of the Snake River (from tributary release sites to Asotin at the head of Lower Granite Reservoir) ranged from 2.1 to 3.2 miles/hr (3.4 to 5.1 km/hr). Migration rates in Lower Granite Reservoir ranged between .9 and 1.1 miles/hr (1.5 and 1.8 km/hr) (Adams et al. 1998). Natural subyearling chinook salmon that were PIT-tagged in the vicinity of river mile 215 in Snake River from 1995 to 1997 took an average of 42 days to pass Lower Granite Dam compared to an average of 41 days for fish that were PIT-tagged and released 50 miles (81 km) downstream (W.P. Connor, USFWS, unpublished data). Fish released upstream probably traveled the 50 miles (81 km) of free-flowing river between the release sites in one day. For comparison, Smith et al. (1997a) reported that it took PIT-tagged hatchery subyearling fall chinook salmon an average of six days to pass through the 38 miles (60 km) of Little Goose Reservoir and by Little Goose Dam. From the above information, it could take up to 22 days for fish to pass through the four Snake River reservoirs compared to only three days if the four lower Snake River dams were breached.

Survival of subyearling fall chinook is likely to be higher than that which is now observed when fish pass through the existing reservoir and dam system. Recent survival estimates for hatchery fall chinook salmon in the free-flowing reach of the Snake River developed by Muir and Peterson (1998) indicate the survival per kilometer for the reach from Pittsburg Landing (within Hells Canyon) to Billy Creek (near the head of Lower Granite Reservoir) was about 0.999 during 1995 and 1997 studies.

4.3.3.3 Rearing and Migration of Juvenile Fall Chinook Salmon

Approximately 139 miles (224 km) of riverine habitat would be restored for juvenile rearing. Fall chinook salmon evolved to rear in riverine habitat. Available information indicates that river-reared parr survive at higher rates than reservoir-reared fish. Smith et al. (1997b) released PIT-tagged subyearling chinook salmon parr in the Snake River at river mile 217 and river mile 166. The parr released at river mile 217 reared primarily in the river, while the fish released at river mile 166 reared primarily in Lower Granite Reservoir. Smolt survival was equal between the above two treatments since fish were released on the same dates and passed Lower Granite Dam over similar time periods. Mortality per kilometer calculated for the time period between release and passage by Lower Granite Dam was almost twice as high for reservoir-reared fish (3.9%/km) than for river-reared fish (2.1%/km). Lower survival of reservoir-reared fish was because of the unnatural rearing conditions in Lower Granite Reservoir. Breaching the four lower Snake River would probably improve rearing conditions and the survival of fall chinook salmon parr in the lower 139 miles (224 km) of the Snake River.

Juvenile salmonids would not be subjected to the extremely high levels of total dissolved gas that can be produced during times of uncontrolled spill at the lower Snake River dams. For example, total dissolved gas levels were above 120% during much of the spring and early summer in 1996 and 1997 and exceeded 130% at times. Hells Canyon and Dworshak dams may also produce high total dissolved gas levels during times of uncontrolled spill. Riverine conditions in the lower Snake River would help to dissipate the high total dissolved gas levels produced by Dworshak and Hells Canyon dams. Data for the Hells Canyon reach show that total dissolved gas dissipated from nearly 126% to 110% in about 30 river miles (48 km) of river and from about 121% to 110% in about 45 miles (72 km) at spills of 13,400 cfs and 21,000 cfs, respectively (Idaho Power Company, unpublished data).

Aquatic invertebrates that now inhabit the reservoirs would be replaced by those found in a riverine environment. With drawdown more riparian vegetation would become established which would increase the production of terrestrial insects. Return to a riverine condition would leave a more complex environment of pools, riffles, and rapids. This would provide a greater diversity of aquatic invertebrates and possibly abundance of food items.

Food habits of juvenile salmonids would change from one based on a reservoir system to one more typical of a riverine system. Before impoundment by Lower Granite Reservoir, the main aquatic invertebrates in that reach of the Snake River were mayflies and caddisflies (Edwards and Funk 1974 in Curet 1993). In the remaining free-flowing reach of the Snake River, fall chinook salmon have been found to feed primarily on mayflies, midges, and caddisflies. Terrestrial insects were found to compose three times the number and four times the biomass of food items eaten by subyearling fall chinook compared to aquatic organisms (K. Tiffan, U.S. Geological Survey, Columbia River Research Laboratory, personal communication). Rondorf et al. (1990) reported that caddisflies which were eaten by subyearling fall chinook in the free-flowing reach of the Columbia River provided from 80 to 160 times the energy content of individual daphnia (waterfleas) that were consumed by these fish in reservoirs. *Daphnia* were the major aquatic organism eaten in McNary Reservoir.

4.3.3.4 Adult Fish Migration

Before construction of the lower Snake River dams, chinook salmon migrated upstream at rates of 12.4 to 14.9 miles (20 to 24 km) per day in the spring and summer. Steelhead migrated at rates of 6.2 to 9.9 miles (10 to 16 km) per day when they were actively migrating in the summer, early fall, and spring. Sockeye salmon migrated at rates of 11.8 miles (19 km) per day (Bjornn and Peery 1992). Since the impoundment of the lower Snake River adult salmonid migration has been altered. Adult fish passage has been delayed at dams, some fish have fallen back over the dams after initially passing them, fish have been exposed to high levels of total dissolved gas during times of uncontrolled spill, and some fish have suffered physical injury such as head burns.

Breaching of the four lower Snake River dams would allow unimpeded upstream migration of adult salmonids to spawning areas in the mainstem river and tributaries. Migration rates and conditions would be similar to those that existed before the dams were constructed and the river was impounded. Adult salmonids would not be subject to delay or fallback during passage at any dams. Fish would also not be exposed to high levels of total dissolved gases that now occur during periods of uncontrolled spill. Head burns and other physical injuries that occur at dams would be eliminated.

Fish passage structures would be placed in the new river channel at the dam breach sites to ensure that adult salmonids would be able to pass these areas. Precast concrete blocks are presently proposed to be placed along the shoreline to create zones of slower velocity (Raytheon Infrastructure, Inc. 1998). These structures would be installed wherever the river velocity is greater than five feet per second so that fish passage velocity criteria could be met. This would allow fish to rest while passing the dam breach sites. The structures would be designed to enable fish passage at a maximum flow of 170 kcfs. They would also be designed to withstand a river flow of up to 420 kcfs.

The biological benefits resulting from restoration of riverine conditions in the lower Snake River could offset the need for flow augmentation during the spring migration period. Summer flow augmentation for fall chinook may not be needed because the net benefit of natural river conditions may be greater than the loss resulting from reduced flows in the lower Columbia River. Restoration of natural river conditions in the lower Snake River may also allow more juvenile fall chinook to migrate during their historical period near the end of the spring freshet rather than during the low flows and high temperatures of summer.

Streamflow augmentation using water from Dworshak and Brownlee reservoirs has been used to increase the downstream migration rate of juvenile salmonids through the lower Snake River reservoirs. It may also be used as a tool to aid the migrations of adult and juvenile fish and spawning of fall chinook salmon under the natural river drawdown condition. Although natural river drawdown would likely increase the rate at which juvenile salmonids pass through the lower Snake River to the Columbia River, flow augmentation may benefit juvenile migrants in dry years when river flows would be low or during the summer migration period when flows typically decline and water temperatures increase.

Flow augmentation using cooler water from Dworshak Reservoir has also been used to improve the survival of summer migrants such as subyearling fall chinook and sockeye salmon during their passage through the lower Snake River reservoirs. Flow augmentation with temperature control may increase the survival of subyearling fall chinook and sockeye during their downstream migration through the free-flowing Snake River after drawdown.

Natural River Drawdown would increase the survival of juvenile anadromous salmonids in the Snake River. However, these fish would still have to pass the four lower Columbia River dams and reservoirs: McNary; John Day; The Dalles; and Bonneville. The NMFS' 1995 and 1998 Supplemental FCRPS Biological Opinions have established spring and summer flow targets for the Columbia River at McNary Dam to improve juvenile fish survival during their passage through the lower Columbia River. Snake River summer flow needs could be met by providing adequate flow from Dworshak and Brownlee reservoirs to help meet the flow target at McNary Dam. The summer flow target is 200 kcfs from July 1 through August 31. This flow would benefit juvenile salmonids from the Snake River as well as those from the Columbia River as they migrate through the lower Columbia River reservoirs.

4.4 Previewing the Key Issues and Technical Intricacies

4.4.1 Overview

Recent (post-1990) smolt-to-adult return rates for threatened salmon stocks appear to be too low to sustain vigorous populations in the face of ordinary environmental fluctuations. In addition, there is no doubt that smolt-to-adult return rates were much higher in the past (prior to 1970) when salmonid populations were also much higher. Scientific complexity arises because there are many environmental factors that have changed over the last century in ways that might have negative impacts on salmon; thus, identifying singular changes that are responsible for salmon declines is problematic (NRC 1996). One way of tackling the problem is to associate past changes with "blame" – in other words identify particular components of the fish life cycle (see

Figure 4.4.1-1) that are negatively affected by particular environmental factors, and then manage for survival and recovery by altering the responsible environmental factors. The idea is simple – to cure a sick person, you have to identify the disease. Unfortunately, although logically appealing, this perspective is very difficult to apply in practice. First, to extend the analogy, the patient's symptoms are consistent with those of many different diseases. In other words, there are many factors potentially affecting the ecological health of salmon populations. For example, the recent NRC report *Upstream* shows graphical plots of salmon declines in the entire Columbia Basin concordant with: human population growth, construction of dams, increased logging, harvest, acres of irrigated lands, and so forth (NRC 1996). Similar correlations exist on the finer scale of Snake River salmon stocks, which are well illustrated by simply displaying the population trajectories or trends in smolt-to-adult returns for spring/summer chinook salmon in conjunction with number of dams (**Figure 4.4.1-2**), total hatchery releases (**Figure 4.4.1-3**), or indices of ocean conditions (**Figure 4.4.1-4**). In addition, the question of blame (and hence "cure") is not something that is easily answered by performing some definitive experiment. The approach adopted by PATH and in this assessment is highly statistical – attempts are made to correlate past declines with changes in environmental factors. But, as the scientific platitude says, "correlation does not imply causation" and the result is the enormous uncertainty discussed in **Section 3.2**. Moreover, it is unlikely that any single factor is responsible for salmon declines; a combination of environmental and human-induced threats has placed salmon at risk (NRC 1996).

Before discussing specific analyses, this section introduces key technical ideas that contribute to the scientific debate surrounding strategies for salmonid recovery, and that require understanding before one can follow particular analyses. To help the reader, a glossary of frequently used technical terms is provided in **Table 4.4.1-1**. Although not all of the terms in this glossary are discussed in this section of the report, this glossary is intended to provide the reader with a convenient reference for terms used throughout the report.

4.4.2 Differential Delayed Transportation Mortality

As discussed in **Section 4.3**, many fish are transported in barges to below the Bonneville Dam (e.g., between 50 and 60% of the spring/summer chinook salmon in 1996 and 1997; Marmorek et al. 1998a). Before they return to spawn, these barged fish may suffer an additional mortality above and beyond what they would suffer if they were not barged; the additional mortality that barged fish may experience below Bonneville Dam is called "differential delayed transportation mortality". It is important to realize that absence of differential delayed transportation mortality would <u>not</u> mean that there was no mortality – rather it would mean that transported fish and nontransported fish suffered the same mortality below Bonneville Dam. The actual process of estimating differential delayed transportation mortality is complicated, but the significance of this mortality, in the context of the PATH analyses, is straightforward. Because differential delayed transportation mortality is a discrete package of mortality associated with the hydrosystem, it is often viewed as an "improvable" factor that can be readily corrected by the removal of dams. However, in reality, even if differential delayed transportation mortality is large, breaching may not be a silver bullet because the delayed mortality could result from transportation of fish that were diseased or in poor condition.

Estimates of differential delayed transportation mortality have been made for outmigration years spanning two decades. These estimates are important because they are used in the prospective PATH simulations in a way that plays a major role in determining the relative value of dam breaching versus leaving the dams intact. Scientists differ in which estimates of differential delayed transportation mortality they feel should be given the greatest credence. The parameter of interest in this debate is the so-called “D-value” (the ratio of survival below Bonneville Dam for transported fish compared to untransported fish, see **Table 4.4.1-1**); D = 1 would mean no differential delayed transportation mortality and a D-value substantially lower than 1 would correspond to high differential delayed transportation mortality (for example a D = 0.33 would indicate that transported fish die at three times the rate as in-river migrants once all the fish are below Bonneville Dam).

4.4.3 “Extra Mortality”

A second important technical concept is “extra mortality”. Time series of adult returns for salmon and steelhead indicate that many stocks declined throughout the Pacific Northwest in the late 1970s (not just stocks on the lower Snake River) (NRC 1996b). However, stocks from the Snake River Basin seemed to decline more than mid-Columbia stocks (which spawn in tributaries that enter the mainstem downstream from the four Snake River dams). Moreover, even after accounting for losses suffered by salmon during their juvenile migration phase (passing downstream through several hydrosystem projects), additional losses must occur to produce the low smolt-to-adult returns seen in many chinook salmon stocks. The unexplained mortality that occurs outside the migration corridor is called “extra mortality”. This is the mortality “needed” to “balance the books” and produce the observed low smolt-to-adult returns, after all other mortality factors have been included in the demographic analyses. Using PIT-tag technology and mark-recapture statistics, it is increasingly possible to quantify mortality through the juvenile migration phase, and hence to know how much “left-over” mortality is unaccounted for and unexplained. However, the cause to which we should ascribe extra mortality remains elusive. Three major collections of hypothesized sources of extra mortality have been examined by PATH: hydrosystem, ocean regime shift, and stock viability degradation. We discuss each of these hypothesized sources of extra mortality, below.

4.4.3.1 Hydrosystem Extra Mortality

Hydrosystem extra mortality includes any effect of the hydrosystem on salmonid survival that is not measured during juvenile downstream migration or adult upstream migration, that does not include differential delayed transportation mortality, and that does not include in-common environmental trends that are reflected in concert in stocks above and below the Snake River dams. A wide variety of mechanisms could produce such an extra mortality. For example, as a result of changes to natural flow conditions, the hydrosystem may alter the timing of fish arrival in the ocean. Or, because of modifications to the river system, the fish may arrive at the ocean in a weakened state that renders them more vulnerable to predation and disease after getting below Bonneville Dam. Changes in the Columbia and Snake River systems have been dramatic, as is described in the FWCAR (USFWS 1998), and such dramatic changes may certainly have yielded a stock of fish less fit for life in the estuaries and oceans.

4.4.3.2 Regime Shift Extra Mortality

A second important subset of extra mortality hypotheses is comprised of the “regime shift hypotheses” or “ocean conditions” hypotheses. These hypotheses attribute the recent low survival of salmonids to changes in ocean conditions. There are many cycles in oceanic conditions that alter patterns of circulation, the distribution of predators and prey, and productivity (NRC 1996b). El Niño fluctuations occur on the timescale of years; Pacific interdecadal oscillations occur on the timescale of decades; other cycles (such as ice ages) appear to operate on timescales of thousands of years. Again, the data are correlational, and the highest correlations are observed for trends that pertain to salmon in Alaska or in Canada (only sparse data are available for the Snake River stocks). But there are strong statistical indications that in many salmon stocks, survival and growth are significantly correlated with changes in the Pacific Decadal Oscillation (PDO) index, a composite index of climatic variation that incorporates the average annual coastal temperature, the average annual basin temperature, and snow depth in March. Over the period of reliable data (1946 to present), the greatest anomalies in sea surface temperatures occurred during the decade 1977 to 1986, coinciding with the onset of low smolt-to-adult return rates for salmon (see **Figure 4.4.1-4** for one picture of climate/stock performance correlations). It is worth noting that the linkage between ocean conditions and salmon performance is not simply a statistical correlation without a plausible mechanism; periods of positive anomalies for the Pacific Decadal Oscillation Index are associated with warm winters and low rainfall that translate into low spring flow rates, which in turn are less favorable for salmonids. The ocean is implicated as a potentially major factor because there are stocks of salmon that do not pass any dams or that come from rivers with no harvest, hatcheries or habitat degradation, yet still have suffered recent declines. One example is steelhead in the Keogh River of British Columbia, which has collapsed from 3,000 adult spawners to 12 adults spawners in the last few years (Welch 1999). The marine survival of Oregon coastal coho salmon was 6.1% from 1960 to 1977, but only 0.6% from 1991 to 1998. These data are not directly applicable to the salmon stocks addressed in this report but they indicate the plausibility of a connection between ocean conditions and salmon performance. Under the "regime shift" hypotheses for extra mortality, different futures are possible depending on how one assumes future ocean conditions will change. If ocean conditions are cycling, then salmon stocks will improve automatically without any management simply because the ocean becomes more favorable. If ocean conditions stay the same or decline, then ocean conditions can mask or limit the ability of management actions to recover stocks.

Whereas it is certainly plausible that ocean conditions influence salmonid performance overall, it requires additional assumptions to explain why Snake River stocks should be "more affected" (hence "extra mortality") than lower Columbia River stocks. This could happen because Snake River and lower Columbia River stocks go to different places in the ocean or because Snake River stocks must travel farther and the extra travel alters their interaction with ocean conditions. With the exception of genetic distinctness, there is a paucity of data germane to these possibilities.

4.4.3.3 Stock Viability Degradation

The third large category of "extra mortality" is stock viability degradation (which is often labeled in PATH documents as the BKD hypotheses, where BKD is an abbreviation for "bacterial kidney disease"). However, degraded stock viability is really something of a catch-all bin for extra mortality. It can represent the effects of many factors including the negative effects (ecological or genetic) of hatcheries on wild stocks, enhanced predation by species exotic to the Columbia River Basin (such as Caspian terns nesting on man-made islands at the mouth of the Columbia River), enhanced diseases, inbreeding depression, and so on. What separates "stock viability" from the other extra-mortality hypotheses is that, unlike the case with regime shift hypothesis, there is no known natural cycle that might work to restore viability; and unlike the case with hydrosystem hypothesis, the removal of dams would not be likely to mitigate this mortality.

4.4.3.4 Assumptions About Extra Mortality Determine Predicted Responses to Management Actions

Management could mitigate certain (but not all) causes of extra mortality. For instance, if extra mortality is due to the fact dams have dramatically altered river ecosystems (the so-called hydrosystem hypotheses for extra mortality), then management that returns the river to more natural conditions is likely to reduce this extra mortality and contribute substantially to recovery of the stocks. However, if extra mortality is due largely to conditions in the ocean, then ocean factors outside our control will constrain management strategies and actions such as dam breaching or habitat improvement may do little to recover the stocks. Thus, without even running a mathematical model, straightforward logic makes clear what the PATH analyses reinforce, as described later in this document – assumptions about the source of extra mortality can govern conclusions about the efficacy of alternative management actions.

4.4.4 Returning to the "Natural River"

The PATH process and NMFS have analyzed the question of salmon survival and recovery by using quantitative models that explicitly treat salmon numbers and link those numbers through widely accepted population models to a variety of management actions. Although there is debate and uncertainty surrounding the interpretation of results from these life-cycle population models, there is wide consensus that the life-cycle models provide a sound currency with which to analyze salmon survival and recovery. But, there is some debate as to whether the analytical approach is too simplistic and restrictive in its view. The argument can be summarized as follows:

It is obvious that the Snake River (and many other rivers in the Pacific Northwest) are drastically altered from their free-flowing, natural condition. Given this observation, is it not equally obvious that removing dams and returning the rivers to their natural condition is the obvious solution?

The "natural river" view is a valid perspective and is ecologically appealing, but implementing this concept in a decision framework is difficult. First, so many changes have taken place over the last century, that it is not possible to restore all of the attributes of the natural river condition (ISG 1996). Thus, the question becomes, *how close to the natural river condition might we move the system?* The "natural river" is a multifaceted ideal, and there are several ways of making a river look more natural– which of those changes towards "naturalness" would do the most to promote salmon recovery? Consider by analogy a "dream house" – a beautiful white colonial mansion with deep green shutters, a large front porch with solid white pillars, interior oak paneling and large Douglas fir beams providing the structural foundation. Now, imagine trying to build that house on a limited budget – what do you cut out? and what are the essential features that get you closest to the ideal? This example is analogous to the salmon dilemma where the "natural river" is an ideal. Thus, NMFS has asked *how much salmon recovery do you get for particular management actions that return the river closer to its natural state.* The NMFS believes that the best way to evaluate river management actions is through salmon demography. In other words, improvements in river conditions (or "naturalness") must be linked to measurable improvements in salmon survival or productivity. Approaches based on "looking like a natural river" run the risk of total failure because, in their pursuit of appearances, they neglect the reality of current demographic factors operating on fish (ocean factors, genetic factors, land-use changes, and so on). This does not mean that NMFS rejects the natural river ideal – indeed this ideal is a rich source of hypotheses about processes needed to maintain vigorous salmon populations. It also is an ideal that suggests likely effects of different management actions. But ultimately, the currency for evaluating actions has to be salmon demography and population dynamics, not the physical attributes of a river alone.

4.4.5 Relative as Opposed to Absolute Probabilities

The distinction between relative and absolute probabilities has already been discussed (**Section 3.2.1**). This is an important enough concept that it warrants reiterating here as a theme that runs throughout the report. It is extremely difficult to estimate a true probability for future ecological events, particularly as they pertain to population numbers or persistence (Ludwig 1999). The "public" and "policy makers" often seem to expect science to deliver absolute probabilities – what is the probability of extinction? what is the expected population size? and so forth. Science cannot deliver these absolute numbers, but careful population modeling can deliver rough estimates of relative probabilities. An example may make this distinction clear. It is impractical to predict the probability that you will get into an accident while driving home from work because the likelihood is so low and so many unpredictable external factors influence that risk (traffic, model and condition of your car, weather, and so on). However, it may be possible to say with some confidence that if it snows, or if there is icy rain, the risk of getting into an accident is increased approximately ten-fold. In short, we can make a prediction about relative probability, even when it is impractical to know the absolute probability. Information about relative probabilities may be "fuzzy" compared to knowing the absolute probability of a 50% chance of getting a heads whenever a coin is flipped, but relative probabilities are still very useful for comparing alternative management actions. Given the uncertainties surrounding salmon population ecology, relative probabilities are the best we can hope for in any scientific analysis. In shorthand, many PATH documents simply use the word "probability"; it should be kept in mind that this always refers to a relative probability.

4.4.6 A Simple Summary

Humans have so changed river ecosystems in the Pacific Northwest that it might seem obvious that dam breaching should provide a tremendous benefit for salmon. But conditions have changed so dramatically on all fronts for salmon that the benefits expected upon dam breaching are not as easy to predict as a casual analysis would indicate. If delayed mortality resulting from transportation is reduced (D made close to 1) over what appears to have occurred historically, and if the hydrosystem is not the source of extra mortality, then dam breaching may, in fact, not markedly improve salmon populations. But, if D-values remain low, or if the hydrosystem is the source of extra mortality, then dam breaching becomes an obviously favored management action. Under less black-and-white mixes of assumptions, the necessity or sufficiency of dam breaching gets muddier, and it becomes necessary to make more detailed comparisons of risk. Although several of the conclusions of this report are clear without sophisticated models, the models are needed to express uncertainties about what is not obvious. When expressing risks and uncertainties, the measures used are all <u>relative</u>. The concept of a relative probability is particularly important: it is useful for comparing alternative actions rather than for predicting exact future conditions.

5.0 Spring/Summer Chinook Salmon

5.1 Population Ecology and Trends

The Snake River Basin includes an area of approximately 107,000 square miles, almost one half of the total area of the Columbia River Basin. Snake River spring/summer chinook salmon are stream-type fish, rearing for a year or more in freshwater before migrating to the sea. After one or more years in the ocean, the adults return to the Columbia River and eventually to their natal tributaries. Returning adults enter the Columbia from early April through July. Some populations return primarily during the spring months, others during the summer. To conduct the analyses, spawner and recruit data were developed for seven Snake River spring/summer chinook index stocks: Minam River (Grande Ronde Subbasin, Oregon), Imnaha River (Imnaha Subbasin, Oregon), Bear Valley/Elk Creek, Marsh, and Sulphur creeks (Middle Fork Salmon Subbasin, Idaho) and Johnson Creek and Poverty Flat (South Fork Salmon Subbasin, Idaho). The Grande Ronde River and Middle Fork Salmon River stocks in this analysis are spring chinook salmon, and the South Fork Salmon River stocks are summer chinook salmon, while the Imnaha River stock has an adult run timing intermediate to those of spring and summer chinook salmon. The numbers of some of these index stocks have fallen precariously low during recent years (**Figure 5.1-1**), indicating that some populations are subject to a high level of extinction risk.

5.1.1 Habitat Trends and Factors

Historically, spring/summer chinook salmon spawned in virtually all accessible and suitable habitat in the Snake River Basin upstream from its confluence with the Columbia River (Fulton 1968). Evermann (1894) reported spring-run salmon spawning as far upstream as Rock Creek, a tributary that enters the Snake River just downstream from Auger Falls, more than 896 miles (1,442 km) from the sea.

The Snake River was probably the major producer of spring/summer chinook salmon in the Columbia River Basin, producing about 39% of the spring chinook and 45% of the total summer chinook salmon run at one time (Mallett 1974). The estimated total production of the Snake River was probably in excess of 1.5 million spring and summer chinook salmon for some years during the late 1800s (Matthews and Waples 1991). The Salmon River alone was estimated to have produced about 44% of the spring/summer chinook salmon entering the Columbia River during the period 1957 to 1960 (Fulton 1968). Adult escapement to the Snake River averaged about 37,100 spring chinook and 22,300 summer chinook from the years 1962 to 1974.

The irrigation and hydropower dams that were built on many of the upper Snake River tributaries eliminated spring/summer chinook salmon from those streams. Irrigation withdrawls, timber harvest and transportation practices, and gold dredging also contributed to the loss of these runs. Barber Dam on the Boise River (1906), Black Canyon Dam on the Payette River (1923), Swan Falls Dam on the mainstem Snake River (1923), Thief Valley Dam on the Powder River (1931), Unity Dam on the Burnt River (1940), Owyhee Dam on the Owyhee River (1933), and Lewiston Dam on the Clearwater River (1927) were among the larger dams in the Snake River system that eliminated native runs of spring/summer chinook salmon. Construction of the Hells Canyon complex of dams during the late 1950s blocked anadromous fish access to the entire upper Snake River Basin.

Quigley and Arbelbide (1997) thoroughly reviewed the extent to which human activity has altered habitat in the Snake River Basin. Logging, agriculture, mining, and urban development have all resulted in a progressive decline in habitat quality. As early as the mid-19th century, grazing of cattle and sheep in the Snake River watershed had altered riparian vegetation, greatly reducing the abundance of trees and shrubs and accelerating bank erosion and channel incision (Elmore and Kaufman 1994). Larger streams and rivers were cleaned of woody debris and other obstructions to aid navigation during the later part of the 1800s, resulting in lower quality spawning and rearing habitat. Complex floodplain habitats were eliminated in many areas by diking, draining, and filling wetlands and ponds and channelizing riparian sloughs and tributaries. In addition to eliminating habitat, these activities (as well as mining and industry) have decreased water quality of some streams in the Snake River Basin (Quigley and Arbelbide 1997).

A second compounding stress that may have implications for spawning habitat quality in the Snake River Basin involves the feedback between returning salmon spawners and nutrient enhancement of aquatic productivity. In general, when salmon die after spawning, the carcasses can represent major nutrient inputs that in turn stimulate productivity. Although relatively little is known about the role salmon carcasses played in the Snake River watershed, research from other systems suggests that such inputs can boost subsequent salmon production (Johnston et al. 1990, Bilby et al. 1996, Bilby et al. 1998). This raises the possibility of a feedback loop whereby any factor that kills salmon prior to their upstream migration will reduce nutrient input and salmon productivity, which in turn exacerbates further salmon declines, leading to further reductions in nutrient input, and so on. Although this scenario has not been pursued in a formal quantitative way, the likelihood that it contributed to the decline of spring/summer chinook salmon is made evident by the fact that salmon biomass deposited in the Snake River watershed had declined 90% from historical levels by the 1960s (**Table 5.1.1-1**).

5.1.2 Hatchery Production

The production of salmonid smolts from Snake River hatcheries (both of spring/summer chinook and steelhead), has increased greatly during the years when naturally-spawned Snake River spring/summer chinook salmon smolts from the 1968 through 1990 brood years were outmigrating through the lower Snake River hydrosystem (Williams et al. 1998a). Most of those brood years yielded low smolt-to-adult return rates for wild stocks (Williams et al. 1998b) (**Figure 4.4.1-3**). Based on the coincidence of these factors in time, NMFS has hypothesized that hatchery production may have had a negative effect on the wild spring/summer chinook salmon (i.e., particularly for brood years 1984 through 1990) through mechanisms related to reduced growth rate, heightened stress, increased predation, and disease transmission (Williams et al. 1998a, Waples 1999). Under this hypothesis, the effects of hatchery interactions are likely to have occurred in the migration corridor, prior to arrival at the first Snake River dam, and were probably exacerbated in areas where fish concentrate (forebays, bypass systems, collection raceways, and barges). Thus, it is reasonable to hypothesize a potential interaction between hatchery production and the concentration and co-mingling of wild and hatchery fish in the transportation program.

The effects of hatcheries are probably greater for Snake River stocks than for mid-Columbia River stocks because:

- The migration corridor prior to arrival at the first dam is much longer for Snake River stocks than for mid-Columbia River stocks, leading to a greater potential for hatchery and wild smolt interactions;

- One of the primary concentrating mechanisms, smolt transportation, is experienced only by the Snake River stock; and

- The natal streams of Snake River stocks are potentially more nutrient-depleted than those of mid-Columbia River stocks, which combined with the more demanding migration of Snake River stocks, would affect fish condition and energy reserves and potentially exacerbate effects of hatchery interactions in the migration corridor.

Retrospective PATH analyses have implicitly incorporated the increase in hatchery production in the Snake River system as a potential causal factor underlying the decline in survival of spring/summer chinook salmon stocks. Under this set of assumptions, the PATH retrospective analyses manifest interactions with hatchery fish as the proportion of spring/summer chinook salmon mortality not attributable to hydrosystem survival or climate effects. In other words, "interactions with hatchery fish" is one possible source of "extra mortality" (and placed in the category of "reduced stock viability"). Research that would more explicitly examine hatchery impacts is outlined in **Section 10.1.4** of this report.

5.2 Adult Harvest and Upstream Passage

5.2.1 Adult Harvest

Historically, a substantial portion of the adult Snake River spring/summer chinook salmon run was harvested in the mainstem of the Columbia River. Snake River runs were harvested in commercial net fisheries in the lower Columbia River and by tribal fisheries above Bonneville Dam. Recreational and tribal fisherman also harvested these stocks in Snake River Basin tributaries. As the runs declined during the 1960s and 1970s, harvest rates were drastically curtailed in the fisheries that affected upriver spring/summer chinook salmon runs. Harvest of wild-origin spring and summer chinook salmon in mainstem fisheries is estimated to have ranged from between 3 and 8% since 1978 (Marmorek et al. 1998).

5.2.2 Upstream Passage

Comparative counts of adult returns passing through ladders at the mainstem dams are used to estimate losses during upstream migration (Beamesderfer et al. 1998). Estimated survival during upstream migration is expressed as a "conversion rate". Conversion rates are calculated by dividing the count of a particular group of adult fish at the uppermost dam by the count of that group at the lowest dam, subtracting out estimates of harvest and tributary turnoff between the dams.

$$Conversion\ Rate = \frac{(Count\ at\ Upper\ Dam)}{([Count\ at\ Lower\ Dam] - [Tributary\ Turnoff] - [}$$

Generally, upstream passage for Snake River fish is divided into two components: passage between Bonneville and McNary dams and passage between McNary and Lower Granite dams. Retrospective estimates of conversion rates for Snake River spring chinook salmon during upstream passage between Bonneville and Lower Granite dams averaged 0.68 for the period between 1977 and 1992. The recent average conversion rate for the four-dam lower Snake River reach was 0.85. To describe the future under different management options, it is also necessary to estimate conversion rates in the absence of the four lower Snake River dams. The retrospective PATH analysis indicated that the most likely upstream survival prior to construction of these dams was 0.97, meaning that dam breaching would be expected to improve conversion rates for that stretch from 0.85 to 0.97.

The conversion rate method of estimating upstream passage survival has a potential bias related to the differential fallback of upstream migrating adults at the dams where counts are made. A detailed discussion of this potential problem, including a comparison of upstream survival estimates made using different methods, is included in **Section 6.2.2**. However, for spring/summer chinook salmon, survival estimates derived from PIT-tag experiments (C. Paulsen memorandum 2/17/99) were similar to estimates based on conversion rates. In addition, the radio-telemetry studies summarized in Marmorek et al. (1998a) indicate a mean project survival estimate for the 4-dam Snake River reach of 0.847, essentially identical to the conversion-rate based estimate of 0.85 for the same reach.

5.3 Egg-to-Smolt Life Stage

The egg-to-outmigrating-smolt stage for Snake River spring/summer chinook salmon covers at least three critical time periods: incubation in the interstices of the spawning gravels, early rearing in the tributaries, and overwintering as juveniles. Egg-to-smolt survival is variable, and we have very imperfect knowledge of the relationship between quantity and quality of habitat and fishery productivity.

Although habitat quality is an important factor in salmon demography, the dramatic collapse of spring/summer chinook salmon populations during the mid-1970s is not correlated with reduced smolt-per-spawner ratios (Petrosky and Schaller 1996). Whereas the annual number of spring/summer chinook salmon returning to spawn declined precipitously in the mid-1970s (**Figure 4.4.1-4**), there was no concordant precipitous decline in habitat productivity as measured by smolts per spawner (**Figure 5.3-1**).

Snake River spring/summer chinook salmon populations spawn and rear in a variety of tributaries within the Snake River Basin. Habitat conditions in those tributaries range from relatively pristine wilderness to drainages that are heavily degraded by human activities. If habitat were a primary factor determining chinook salmon population declines in the Snake River, then the trend in returns should differ among tributaries with differing habitat conditions. However, the recent downward trend in returns is generally similar among stocks originating in areas with markedly different habitat conditions (Marmorek et al. 1996). Although habitat conditions may not explain yearly fluctuations in smolt-to-adult return ratios, they could still be crucial to a stock's long-term productivity and viability. NMFS believes that more basic research should be aimed at linking habitat attributes to productivity (see **Section 10.1.3**).

5.4 Smolt-to-Adult Life Stage

Estimates of smolt-to-adult return (SAR) rates (**Figure 4.4.1-4;** lower graph) indicate that survival has dramatically declined over the last 30 years (Marmorek et al. 1998b, Marmorek and Peters 1998b). Clearly, mortality in the smolt-to-adult life stage plays a major role in the observed, parallel decline in adult returns.

Estimates of survival through the different components of this complex and extended life-history phase are difficult to obtain. In general, the PATH process has broken survival into two categories:

- Direct survival of outmigrating fish from the head of the hydrosystem to below Bonneville Dam and

- Survival from below Bonneville Dam until the fish return to their natal streams as spawning adults.

Detection of fish at dams during upstream passage provides a means of estimating conversion rates (**Section 5.2.2**). Thus, the major unknown factor is survival in the estuary and ocean. The PATH analyses break estuary and ocean survival into three major categories:

- Mortality due to climatic fluctuations, which are felt uniformly across all stocks (regardless of whether they are above or below the Snake River dams);

- Differential delayed transportation mortality, which is experienced only by transported fish (**Section 4.3.2**); and

- Extra mortality, or the unexplained mortality affecting Snake River stocks after other factors have been eliminated (**Section 4.3.3**).

In the following sections, we discuss direct survival for spring/summer chinook salmon, then in-common survival associated with climatic fluctuations, differential delayed transportation mortality, and lastly, extra mortality. The level of detail in this discussion for spring/summer chinook salmon is far greater than is possible for fall chinook salmon, sockeye salmon, or steelhead (for which data are much more scarce or analyses from PATH are preliminary).

5.4.1 Direct Survival to Below Bonneville Dam

Mainstem passage survival to below Bonneville Dam has been estimated from fish-marking experiments. Estimates for the historical period, including impacts during years of construction and operation of the Snake River dams, are based on extrapolations from studies over particular reaches within the system. Until recently, it was not possible to estimate survival through the entire mainstem from the upper most Snake River project (i.e., Lower Granite) to below Bonneville Dam. Fortunately, the installation of PIT-tag detectors at Bonneville Dam, combined with the development of trawl-mounted detectors for use in the river below Bonneville Dam, may enable researchers to develop survival estimates over the entire reach. However, at this point in time, detection rates at Bonneville Dam are relatively low and trawl-mounted PIT-tag detectors are still in a developmental stage.

The PATH process developed historical estimates for the mainstem migration by comparing estimates derived from two passage models to reach survival studies as well as to independent estimates of passage survival at some mainstem dams. Primary input into these models included run reconstructions as described by Beamesderfer et al. 1998, and discussed in **Section 3.3**. Each passage model incorporates assumptions regarding dam passage and reservoir survival, and each reflects historical information on smolt migration speeds and timing. Passage through a dam can take three avenues: spilling over the dam, going through the turbines, or bypassing the dam. An alternative route is transportation (via truck or barge). The details of how fish are assigned to these different routes and what mortalities are associated with each route comprise the so-called "passage models" (CRiSP versus FLUSH, see glossary in **Table 4.4.1-1**). For a full exposition on the differences between these models, consult the PATH reports for fiscal years 1997 and 1998 (Marmorek and Peters 1998b, Marmorek et al. 1998a). The passage models estimate survival of the total population of fish from the head of Lower Granite Reservoir to the tailrace of Bonneville Dam. It is noteworthy that, although the passage models differ in assumptions about reservoir mortality, they produce similar estimates of direct survival to below Bonneville Dam under historical conditions. Discussions in PATH documents have often emphasized the uncertainty reflected in choosing either CRiSP or FLUSH as the appropriate models. NMFS believes that the critical difference between the two passage models is the way they estimate D-values (differential delayed transportation mortality). NMFS recognizes that, if FLUSH and CRiSP were forced to run with identical D-values, the models would generate very similar predictions. Hence, instead of belaboring details about CRiSP versus FLUSH, this document focuses on the uncertainty surrounding D-values, and the implications of that uncertainty.

Biologically, the important point about spring/summer chinook salmon direct survival is captured in **Figure 5.4.1-1**. Direct survival to below Bonneville Dam declined sharply in the late 1960s and early 1970s. This decline in migration survival parallels the decline in smolt-to-adult returns and the collapse of spring/summer salmon stocks. However, with subsequent improvements in the hydrosystem (better transportation and bypass facilities) during the 1980s, direct survival to below Bonneville Dam has increased markedly (**Figure 5.4.1-1**). However, smolt-to-adult returns have not increased in parallel with the improvements in direct survival. Hence, it is clear that some additional factors must be keeping SARs undesirably low for spring/summer chinook salmon. The other candidates, to which we now turn, are differential delayed transportation mortality (**Section 4.4.2**) and "extra mortality" (**Section 4.4.3**).

5.4.2 General, In-Common Climate Effects and Smolt-to-Adult Return Rates

Survival through the estuary and ocean life-history phase is affected by year-to-year variation and multi-year trends in climate and environmental effects. The specific mechanisms resulting in patterns in marine survival are not understood. However, several mechanisms underlying these climatic effects are under investigation. For instance, shifts in ocean climate are known to alter rates of primary and secondary productivity, the availability of alternate prey, and the abundance and distribution of predators. Changes in any of these factors will affect ocean survival and smolt-to-adult return rates. The effects of climate change on salmon survival is a vigorous area of research. Among the more unambiguous trends is a major upward shift in smolt-to-adult survival in the mid-1970s for many salmon runs returning to rivers in Alaska and British Columbia (e.g., Beamish and Buillion 1993, Francis and Hare 1994). McGowan et al. (1998) have related these changes in SARs to plankton productivity. Historical catch records for salmon fisheries off Alaska and British Columbia support this hypothesis. For those stocks, the oceanographic regime shift in the 1970s represented the most recent in a series of relatively long-term cycles in ocean/climate effects, each with a period of approximately 30 years (Mantua et al. 1997). At the same time that Alaska and British Columbia stocks experienced an upward shift in SARs, some stocks returning to river systems in Washington and Oregon showed a decline in survival (Mantua et al. 1997). However, the statistical correlations between ocean conditions and survival estimates for the spring/summer chinook salmon stocks returning to the Columbia River are weak (Marmorek et al. 1998a). Instead of assuming one particular linkage between ocean condition and spring/summer chinook salmon demography, PATH explored a range of assumptions for retrospective analyses and used different scenarios for prospective future simulations, as described below and in **Section 5.5.1.4** and **5.5.3.3**.

The PATH analyses indicate that the decline in smolt-to-adult survival of Columbia River stocks in the late 1960s and early 1970s coincided with a downturn in estimated marine survival for spring/summer chinook salmon migrants from natal tributaries both above and below the hydroprojects. The PATH retrospective analyses estimated the contribution of climate and other environmental conditions to the patterns in survival of Snake River spring/summer chinook salmon using two approaches. In the first approach, PATH estimated in-common year-to-year variation in survival among genetically distinct stocks, and attributed this shared variation to ocean conditions. A second approach assumed *a priori*, a relationship between the ocean survival of Snake River spring/summer chinook salmon and indices of ocean conditions (Ocean Station PAPA) and estuarine conditions (Astoria Flow Index). Details can be found in Marmorek et al. (1998a,b).

The PATH process has concluded that the comparative spawner/recruit analysis supports a common pattern in ocean survival for upstream and downstream spring chinook salmon stocks with similar life-history patterns (Marmorek et al. 1997, 1998b). The downstream spring chinook salmon runs used in the comparison (i.e., John Day River, North Fork John Day River/Granite Creek, and Warm Springs River, Oregon, and Klickitat River and Wind River, Washington) show relatively high SARs during the mid-1980s followed by a return to lower survival rates that continue to the present. During 1989 through 1990, a major shift in ocean survival conditions has been hypothesized, based on a common downward shift in survival for many stocks of steelhead and coho salmon returning to river systems in British Columbia, Washington, and Oregon (Welch et al., in press). The decreased recent survival rates observed for steelhead and coho salmon stocks (both species with freshwater life-history patterns similar to those of Snake River spring/summer chinook salmon) coincide with the spring/summer chinook salmon's strikingly low SARs of 1992 and 1993. However, we cannot necessarily infer a similar ocean-based survival for spring/summer chinook salmon as for coho salmon and steelhead, because we do not know whether the species occupy similar ocean habitats.

An important source of uncertainty pertaining to ocean conditions arises when considering options for simulating the future. For example, when simulating future possible salmon trends, it is not clear whether the current downward shift in ocean conditions will persist or perhaps reverse itself. In general, such complicated patterns and scales of climate change make prospective simulations tenuous. PATH's approach to this uncertainty has been to simulate future scenarios using several different climate hypotheses. These simulations to date have not included ocean conditions that become even more unproductive, a possibility that needs consideration. Because future scenarios have neglected ocean conditions that remain poor or become worse, the recovery and survival of simulated populations is optimistic with regard to ocean effects.

5.4.3 Measured Effects of Hydrosystem Passage on Smolt-to-Adult Returns

5.4.3.1 Differential Delayed Transportation Mortality

The D-values employed in PATH analyses to date were derived mostly from transportation studies conducted during the 1970s and 1980s and from estimates of survival for downstream-migrant fish under historical hydrosystem conditions. In the PATH life-cycle model, the D-values represent the survival of transported fish after they leave Bonneville Dam relative to the (post-Bonneville) survival of fish that arrived in the Bonneville Dam tailrace after migrating downstream through the entire hydrosystem. The PIT-tag data discussed below suggest that D-values derived from the transportation program as presently implemented, and current survival conditions for downstream migrants within the hydrosystem, may be higher than the average D-values used to date by PATH.

NMFS used data derived from fish PIT-tagged as juveniles during 1994 and 1995 (years for which relatively large numbers of spring/summer chinook salmon were PIT-tagged and for which adult returns are now complete) to estimate D-values. To construct transported and downstream groups from PIT-tagged fish, NMFS used only PIT-tagged fish with the same passage history as the non-tagged fish in the run-at-large. This was a simple procedure for the transported group; transported PIT-tagged fish represented transported non-tagged fish collected at the same location. However, because most of the non-tagged fish that entered a bypass system at a collector project were transported, the group of fish in the general population that remained in the river all the way downstream to Bonneville Dam passed the dams mainly via spill and turbine routes. Thus, PIT-tagged fish detected (bypassed) multiple times were not considered representative of the downstream group.

NMFS has developed methods to estimate the number of PIT-tagged fish that used each of the possible passage routes during their migration (Sandford and Smith, manuscript submitted for publication). NMFS used these methods to estimate the number of PIT-tagged juvenile fish that survived to the tailrace of Bonneville Dam using passage routes representative of non-tagged downstream migrant fish. In 1994, nearly all of the non-tagged fish that entered the bypass systems at Lower Granite, Little Goose, Lower Monumental, and McNary Dams were transported. Thus, the PIT-tagged fish that best represented the non-tagged fish that survived to Bonneville Dam were those in the “never-detected” group. During the 1995 migration, however, the collection system at McNary Dam operated in “full bypass mode,” returning all fish (tagged or non-tagged) that entered the bypass system to the river. Thus, for 1995, the PIT-tagged fish that best represented the general population of downstream migrants below Bonneville Dam included the “never-detected” group as well as those PIT-tagged fish that were detected and bypassed only at McNary Dam.

Adult returns of wild spring/summer chinook salmon from juveniles PIT-tagged above Lower Granite Dam during the 1994 and 1995 juvenile outmigrations were not sufficient to obtain separate, reliable estimates of D-values. To decrease the variability inherent in estimating the mean, the 1994 and 1995 wild fish were combined to obtain an estimated D-value of 0.81 (“boot-strapped” 95% confidence interval = 0.34 - 1.51).

During the 1995 migration, a substantial number of fish were PIT-tagged at Lower Granite Dam for a study of transportation. Inspection of adult returns from the study indicated that fish released in the tailrace of Lower Granite Dam and subsequently never detected had return percentages comparable to those for fish tagged above Lower Granite Dam and never detected within the hydrosystem. Return percentages were similar for other, comparable passage history categories, so we combined fish tagged at and above Lower Granite Dam, allowing separate estimates of D-values for hatchery and wild fish with smaller ranges. The estimated D-values for from 1995 PIT-tagged hatchery and wild spring/summer chinook salmon were 0.91 (boot-strapped 95% confidence interval = 0.72-1.16) and 0.82 (boot-strapped 95% confidence interval = 0.55 - 1.18), respectively.

The more recent estimates of D-values are higher than those used in prospective analyses by either CRiSP or FLUSH passage models. The mean D-value for CRiSP is 0.66, whereas the mean D-values for FLUSH vary from 0.31 to 0.53. Both of these sets of mean D-values are clearly lower than the D-values estimated from the recent PIT-tag experiments. However, it is important to note that the 95% confidence intervals for the recent estimate of D = 0.81 are large, and that these data represent findings from only two outmigration years. A larger sample size is needed to reduce the sampling error, and more years of data are needed to span a broader range of environmental conditions. There is scientific debate surrounding how much weight to place on these most recent D-estimates. NMFS scientists believe these PIT-tag results should be given substantially greater weight because the method of estimation is much improved over past methods and because they better reflect current operations. An alternative view places great weight on D-values derived from historical data because more years are involved in garnering those estimates (and hence a wider range of environmental conditions are sampled). Because both perspectives have merit, this report presents results for a range of D-values.

5.4.3.2 Effect of Bypass Systems

The same PIT-tag experiment used to estimate D-values (**Section 5.4.3.1**), can also be used to examine the SARs of fish that pass through different numbers of bypass systems. In particular, the fish in the in-river group were released into the tailrace at Lower Granite Dam and passed downstream through zero, one, two, or three additional juvenile collection facilities at Little Goose, Lower Monumental, and McNary Dams. The PIT-tag detection records at these facilities and those at Lower Granite Dam for returning adults were used to estimate SARs for subgroups of fish aggregated according to the number of times they were detected. For wild fish, the SARs were 0.23% (15 of 6,544), 0.18% (22 of 12,512), 0.13% (9 of 6,801), and 0.19% (3 of 1,602), respectively, for fish detected zero, one, two, or three times after leaving Lower Granite Dam (**Figure 5.4.3.2-1**). With the exception of the SAR based on only three returning fish (fish detected three times), there appears to be a trend towards increased SARs for groups of fish that passed through fewer bypass systems as juveniles. Larger sample sizes and more years of data collection are needed to test this hypothesis.

5.4.4 Extra Mortality

“Extra Mortality” is defined as any mortality of Snake River salmon and steelhead that occurs outside of the juvenile migration corridor and is not accounted for by either: productivity parameters in spawner-recruit relationships, estimates of direct mortality within the migration corridor (from passage models), differential delayed transportation mortality, or common-year effects influencing both Snake River and Lower Columbia River stocks (Marmorek et al. 1998a).

In the context of PATH, extra mortality was estimated as any mortality not accounted for by other terms in the life-cycle model (see **Attachment A** to this report). Specifically, the models were fit to data such that Ricker spawner-recruit parameters were obtained, direct mortality was estimated, environmental variation that simultaneously affects both Snake River and lower Columbia River stocks was determined, and random effects specific to each stock in each year were estimated. Any temporal trend in the residuals (e.g., unexplained variation not assignable to the other model factors) is called extra mortality.

Although the cause of the extra mortality is uncertain, three general factors hypothesized to have contributed to this mortality are:

- Climate/environmental trends specifically affecting Snake River salmon runs;
- Effects of factors other than climate or the Snake River dams (generally referred to as declines in stock viability); and
- Delayed effects of hydrosystem passage (not encapsulated in differential delayed transportation mortality).

5.4.4.1 Climate Regime Shift Hypothesis

A long-term, cyclical shift in climate regime with a period of 60 years has been hypothesized to explain patterns in the extra mortality of Snake River spring/summer chinook salmon. Under this "Regime Shift" hypothesis, effects on the survival of Snake River spring/summer chinook salmon are hypothesized to have changed from positive to negative circa brood year 1975. The climate regime is hypothesized to return to an above-average (favorable) condition starting with brood year 2005. The regime-shift effect on extra mortality would be in addition to any cyclical climate impacts affecting both upriver and downriver stocks in common. The regime shift hypothesis offers an optimistic view for Snake River salmon, because it conjectures that conditions for the fish will improve without any management intervention, simply because the ocean will cycle back to favorable conditions within 5 to 10 years.

5.4.4.2 Reduced Stock Viability

It is possible that the viability of Snake River stocks declined after the early 1970s. This hypothesis states that at least a portion of the mortality below Bonneville Dam does not result from passage through the hydrosystem or from climate conditions. The mechanism originally proposed to explain decreased stock viability was that hatchery programs implemented after construction of the Snake River dams led to an increase in either the prevalence or the severity of bacterial kidney disease (BKD) within the wild population. As a result, it was hypothesized, the mortality of juvenile fish increased after they exited the hydrosystem compared to mortality observed in earlier years.

An alternative mechanism has been proposed involving stress due to interactions between migrating wild Snake River chinook salmon and the large numbers of hatchery-released fish in the system. Hatchery production of chinook salmon and steelhead within the Snake River Basin has increased dramatically in recent years. Evidence from laboratory and field studies supports the assumption that interactions with hatchery fish, in particular large steelhead smolts, can lead to increased stress in spring/summer chinook salmon smolts. The increases in hatchery production were primarily instituted as mitigation for construction of the mainstem Snake River dams (Lower Snake Compensation Plan) or for the effects of construction and operation of the Hells Canyon complex of dams, upstream of Lower Granite Dam.

A third route by which stock viability might decline involves genetic degradation. Foremost among the mechanisms underlying genetic deterioration is the introgression of genes from hatchery fish and a resulting decline in the fitness of wild fish. Other mechanisms include depletion of genetic diversity and inbreeding depression. Such genetic degradation is expected in theory whenever populations become "too small", although what constitutes "too small" is difficult to specify because it depends on so many additional factors (e.g., rate of population growth, dispersal, variation among females in reproductive rates, and so on). Genetic degradation would be gradual and would include a timelag after populations initially fell to dangerously low levels.

The reduced stock viability hypothesis also encompasses the potential that extra mortality is the result of other changes in the estuary or nearshore ocean. For example, the construction of major hydroprojects on the mainstem Columbia River, culminating in the 1970s, has resulted in significant shifts of outflow away from the spring freshet. The Columbia River plume has a major influence on the physical oceanography of the nearshore zone although there is little available information on the effects of changes in the plume on biological processes. A change in predation pressure could also be hypothesized to explain "extra mortality" below Bonneville Dam. A large population of Caspian terns now nests near the mouth of the Columbia River and is estimated to consume between 5 and 30 million smolts annually. These terns were not present in the estuary prior to the mid-1980s. Other predators, such as marine mammals, have also experienced recent population increases with potential consequences for salmon mortality. Salmonids from the Snake River might be more susceptible to predation than Columbia River fish either due to genetic differences or to the added stress of their longer migration (independent of the additional number of dams they must pass).

5.4.4.3 Hydropower Hypotheses Regarding Extra Mortality

Under the hydrosystem extra mortality hypothesis, delayed mortality of Snake River spring/summer chinook salmon is directly associated with the impact of the four lower Snake hydroprojects. If the hydropower extra mortality hypothesis proves to be true, removal of the four dams could potentially return SARs to the higher levels seen in the 1960s (3 to 5%) and hence to substantially promote the recovery of these stocks. The mechanisms by which the hydrosystem could influence survival below Bonneville Dam generally entail "extra stress" or a "weakened condition". The hypothesis is simple – because the river has been so dramatically altered and fish migration is potentially more stressful, the fish entering the ocean are not as "vigorous" as they would be if they did not have to proceed through the hydrosystem.

5.5 Analysis of Hydrosystem Management Alternatives

5.5.1 Future Effects of the Hydrosystem Management Actions

The PATH process, using each of the two alternative passage models, CRiSP and FLUSH, projected juvenile passage survivals under each of the alternative future system options. Alternative sets of assumptions regarding passage parameters were drawn as inputs. The passage models were used to create a series of projected juvenile survivals for each management action corresponding to the range of environmental conditions associated with the historical series (1977 through 1992 migration years), described above. The results are expressed as a series of adjusted in-river survival values for use in the life-cycle analyses described above.

For completeness, a large number of assumptions and modeling details are outlined in this section, giving the impression of a very complicated "story". It is useful to keep in mind that, in the end, it is assumptions about "extra mortality" and differential delayed transportation mortality that largely determine the results.

5.5.1.1 Assumptions Used in Simulations of Future Conditions

In-River Survival

Using the passage models, projected survival rates for in-river migrating juvenile spring/summer chinook salmon were generated for each of the modeled years. Two sets of parameters were used as input to the prospective assessment of in-river survival: dam passage elements and reservoir passage/survival studies. The same elements used in assessing retrospective passage survivals are incorporated into the prospective modeling. Spill levels were set depending on the particular future management option being assessed. Spill survival was assumed as 98%. Alternative assumptions regarding fish guidance efficiency (FGE) and survival while passing through turbines were incorporated into the sets of different assumptions used when producing a series of runs for each management option (**Table 5.5.1.1-1**).

Reservoir Survival

The two passage models use different strategies to project reservoir survival estimates for the spring/summer chinook salmon. The CRiSP model generates survival estimates for reservoir passage using assumptions regarding travel time and hypothesized mortality rates as a function of the time of exposure to predation and to total dissolved gas levels (Appendix A in Marmorek and Peters 1998a). The CRiSP model estimates daily reservoir mortality as a function of temperature. Because water temperatures tend to increase over the spring migration season, predation rates projected by CRiSP show a corresponding increase. The FLUSH model estimates prospective reservoir survival using a set of mathematical relationships based on fish travel time. In particular, for each year modeled, a declining exponential function was used to relate reservoir survival rate to cumulative travel time.

For the preliminary decision analysis, PATH explored two alternative hypotheses. Hypothesis one states that the predator removal program (i.e., removal of northern pikeminnow for rewards) would have no effect on reservoir mortality. Hypothesis two states that predator removal would result in a 25% reduction in reservoir mortality. These two values were chosen to represent the extreme bounds for probable effectiveness of predator removal.

Transportation

For those potential actions that include transportation of smolts, the simulations require three types of assumptions: the set of rules employed to calculate the proportion of migrants collected and transported, an estimate of the survival of smolts during the process of transportation, and an estimate of differential post-Bonneville delayed mortality for transported fish (compared to in-river migrants) that takes effect after the smolts arrive below Bonneville Dam. The fish guidance efficiencies used in the passage models and the rules for spill and collection determined the proportion of fish transported. The FGEs represent the proportion of smolts headed for turbine intakes that are guided by special screens into a bypass/collection system. Estimates of FGE for each dam have been standardized amongst the passage models. Both FLUSH and CRiSP assume that direct survival of transported fish from the point of collection in the bypass system to release below Bonneville Dam is 98%.

FLUSH versus CRiSP Approaches to Differential Delayed Transportation Mortality

Differential delayed transportation mortality is quantified by the ratio of post-Bonneville Dam survival for transported smolts divided by post-Bonneville Dam survival for non-transported smolts. Clearly, this is an important parameter when evaluating drawdown (e.g., alternative A3) as an option because, if D is low, removing dams can increase fish survival (and remove the need for transportation). Conversely, if D is high (e.g., equal to 1.0), then breaching may provide little or no improvement over transportation. The FLUSH and CRiSP models generate estimates of past D-values differently and also draw D-values for prospective future scenarios differently. The details of the methodology involved in these estimates can be found in Marmorek et al. (1998a). For the purpose of this report, it is important only to note that a wide range of assumptions about D was used in the PATH process. The most important distinction between FLUSH and CRiSP is that they ran prospective simulations with different ranges of D-values.

Drawdown

Two drawdown (dam breaching) alternatives were analyzed through the PATH process. One alternative (A3) incorporates the natural river drawdown (breaching) of four Snake River mainstem reservoirs (Lower Granite, Little Goose, and Lower Monumental dams). The second alternative (B1) involves a combination of natural river drawdown of John Day dam on the mainstem Columbia River with the four-pool Snake River option. Modeling the drawdown options involved assumptions regarding four time periods:

- Pre-removal - the period between when the region decides to proceed with drawdown and when physical removal of dams begins.
- Removal – the period in which engineering work to breach or circumvent the dams is carried out.
- Transition – the period beginning just after the dams are removed and continuing until fish populations attain some equilibrated conditions.
- Equilibrium – the period of time from when fish populations equilibrate to the end of the simulation period.

For each period, the PATH process requires assumptions about the duration of the above four periods and estimates of the adult and juvenile survival rates that are expected (**Table 5.5.1.1-2**). The potential for increased juvenile mortality associated with the transition following drawdown was considered in a set of PATH sensitivity analyses (Marmorek et al. 1998a). Two scenarios were considered, decreasing in-river survival for the first 5 years after drawdown by 10% and by 50%, respectively. The 10% and 50% values were not associated with any particular mechanism, but were chosen to provide insight into the potential response to a wide range of possible effects. A limited set of analyses was done using the CRiSP model in combination with best-case passage assumptions and worst-case drawdown assumptions. The results indicated that assumptions regarding juvenile mortality during the transition period had relatively small impacts on the survival and recovery projections.

PATH has identified the need for further analyses of transition and removal effects under a wider range of aggregate assumptions. As can be seen from **Table 5.5.1.1-2**, the removal effects due to breaching do not include any impacts on juvenile or adult survival; the general types of effects that might occur for all salmonids are discussed later in **Section 10.3**. Additional assessments should include a more explicit consideration of extinction risks at extremely low population sizes. Strategies to minimize transition risks should be more completely developed for future analyses.

The alternative drawdown scenarios (A3 and B1) use the same equilibrated juvenile survival rate (equal to a survival rate of 0.85 over the reach corresponding to the four Snake River projects) and the same 3-year preremoval period, but differ in the length of the transition period between dam removal (completed in 2004 in this scenario) and equilibrated levels. In these examples, a regional decision would be made in 1999 and removal of dams would take place between 2002 and 2004. Additional variations involving alternative scenarios for John Day drawdown were run as part of the assessment of action B1.

The transition period is defined as the period of time between the end of the construction period and when the free-flowing river would attain some equilibrium survival rate for juveniles. Physical processes during this period would probably include increased water velocities (reduced travel times), formation of a new channel, washout of accumulated sediments, stabilization of banks, and re-establishment of riparian areas along side the new channel. Biological processes would probably include changes in ecological communities. With respect to the effect of drawdown on juvenile survival rates during the transition period, changes to the density, abundance, activity, and distribution of predator species in the free-flowing river are the primary biological factors under consideration. The response of juvenile survival rates during the transition period is thought to be primarily a function of three processes:

- The response of predator populations to the change from reservoir to free flowing conditions. Specifically:

    - lower water volumes may reduce predator carrying capacity (although initial increases in density are possible);
    - increased turbidity and decreased temperature may reduce consumption rate; and
    - changes in channel morphology and microhabitat distribution may affect distribution of predators and juvenile chinook salmon, which would affect encounter rates;

- The decreased fish travel times that result from increased water velocities reduce exposure of juvenile chinook salmon to predation; and

- The possible direct effects of increased suspended sediments, and of contaminants adsorbed to sediments.

The increase in water velocities under drawdown is generally accepted. The key question, therefore, is whether predator population dynamics will change sufficiently to counteract the positive effects of reduced travel times. A very limited amount of information is available on predator densities and predation rates in free-flowing sections of the Snake River (upstream of Lower Granite Dam) and the Columbia River (below Bonneville Dam). At both study sites, predator densities and consumption rates were higher than in mid-reservoir samples, but the applicability of these data to a free-flowing Snake River is tenuous, and the "data for making broad conclusions are sparse" (review in Petersen and Poe 1998). Work is currently underway to study the effects of plausible habitat changes on predator densities and consumption rates.

Projected Juvenile Survival

The combined effect of the in-river passage assumptions on expected survival under the alternative Snake River hydrosystem actions can be expressed in terms of two aggregate measures: total survival and system survival. Total direct survival is a composite estimate, incorporating the estimated survival of both in-river and transported migrants. Both CRiSP and FLUSH models project relatively high estimates of total direct survival for the future under actions A1 and A2, reflecting the high proportions of the run transported. The projected estimates of direct total survival to below Bonneville Dam for Actions A1 and A2 exceed the corresponding juvenile survivals under action A3, Snake River drawdown, under both modeling systems.

Estimates of system survival for in-river migrants under each action incorporate the differential delayed mortality of transported fish derived as described above. Both the CRiSP/T3 and FLUSH/T1 modeling systems project that system survival under drawdown would exceed system survival under the transportation options. Sensitivity analyses (Appendix D in Marmorek and Peters 1998a) indicate that the different methods of projecting differential transport mortality used by the respective modeling systems account for almost all of the differences in projected survival between CRiSP/T3 and FLUSH/T1.

5.5.1.2 Life-Cycle Modeling

A Bayesian life-cycle modeling framework was developed to carry out the prospective modeling (Deriso 1998). A detailed mathematical description of the model is included as an attachment to this assessment. As was the case with the retrospective analysis, the prospective Bayesian Simulation Model (life-cycle model) is based upon an analysis of the spawner-recruit series for the seven index stocks described in **Section 5.1**. The stock-recruit framework assumes a basic Ricker model with provisions for depensatory mortality at low spawner levels. The results of the modeling are displayed as estimates of the relative probability of stock survival and recovery for comparison with the NMFS criteria described in **Section 3.2.1**.

The life-cycle model was structured to allow incorporation of the assumptions and results from the alternative (i.e., Alpha and Delta) life-cycle models and passage models (CRiSP and FLUSH). The Alpha and Delta models are described briefly in the attachment to this report and more fully in Appendix A.3.2 in Marmorek and Peters (1998a). Briefly, the Delta model is an extension of the model used in Chapter 5 of the PATH Retrospective Analysis (Deriso et al. 1996). Deriso et al. used spawner-recruit data from Snake River and lower Columbia River stocks to infer common-year climate effects shared among all stocks, as well as a combined "direct plus extra" mortality. The prospective Delta model separates the direct and extra mortality components by estimating direct mortality using a passage model while keeping the common year effects as a separate term. Under the Delta model assumptions, the life-cycle model incorporates common year effects, hypothesized as common effects of ocean and climate factors on upriver and downriver stocks with similar life history patterns (but unknown ocean migration patterns). The common year effect was derived from the retrospective analysis and incorporated information for brood years 1952 through 1989. Interestingly, sensitivity analyses indicate that the version of life cycle model chosen (Alpha versus Delta) has negligible effect on the results (see Section 9.2 and **Table 9.2-1**)

The Alpha model also uses a passage model for the direct component, but does not estimate common-year effects based on similarities between Snake River and lower Columbia River stocks. Instead, the Alpha model treats each stock group independently, with an extra mortality term specific to each group that includes both climate effects and any delayed effects of the hydrosystem. Annual variations in climate/environmental effects on ocean survival are incorporated into the Alpha model mathematically.

Within the life-cycle model, the effects of alternative actions on juvenile passage were implemented through a mechanism based on the detailed retrospective modeling of passage survival during the outmigration years 1977 to 1992. The potential change in survival under a given action was calculated for each year in the series using the passage models. The resulting series of projected survival rates was then used in the forward simulations through a two step process. The individual estimates corresponding to the years 1977 to 1992 were assigned a probability based upon the frequency of similar water years in the 50-year record. The revised survivals estimates were drawn based on those probabilities in the prospective model runs.

5.5.1.3 Results of the Decision Analysis

The results of the PATH analytical work conducted to date have been summarized in a series of reports. The following summary draws from those reports, incorporating NMFS scientific conclusions and recommendations. It is important to note that the results reflect only the range of assumptions considered within the PATH process. Potential future actions outside the hydrosystem have not been fully addressed by the PATH process to date. For example, reductions in hatchery releases are not considered. However, sensitivity analyses do allow some insight into the potential impact of alternative harvest schedules, tributary habitat improvements, and different scenarios for variation in ocean conditions.

What alternative management actions most robustly meet performance criteria?

Based on the PATH analyses conducted to date, one can compare the results of alternative hydrosystem actions across all of the potential future conditions reflected by the alternative assumption sets. Actions that meet or exceed survival and recovery benchmarks for a broader set of future alternatives are considered more robust than actions that meet criteria under fewer future assumptions.

The result of a particular combination of alternative assumptions is expressed in terms of the probability of exceeding the survival threshold or recovery levels under that set of assumptions. To incorporate the effect of uncertainties, 4,000 100-year replicate Monte Carlo simulations were run for each set of assumptions. Statistics accumulated across the simulations for each set are used to calculate probabilities relative to survival and recovery escapement levels. In **Table 5.5.1.3-1**, the average probability of exceeding these escapement levels is summarized for each of six alternative management actions.

Table 5.5.1.3-1 indicates clearly that dam breaching (either A3 option) averages an 82% relative probability of meeting recovery population escapement criteria, whereas no-breaching averages a 47 to 50% chance of meeting the recovery criteria. Thus, breaching provides an additional 30% chance of meeting recovery criteria and is hence the most robust or risk-averse option. Differences among hydrosystem actions with respect to survival criteria are not as dramatic (but the differences are in the same direction as those for recovery criteria, with breaching the more robust or risk-averse option).

One problem with reducing the analysis to a single number for each management action (the average probabilities shown in **Table 5.5.1.3-1**), is that a single number does not give information about the variability in the results. Box and Whisker" figures help display this variability. In a "Box and Whiskers" diagram, the upper and lower vertical lines ("Whiskers") represent the range of results across all combinations of the assumptions considered in the quantitative PATH analysis. The "Box" illustrates the range of probabilities associated with the middle 50% of outcomes. An approximation of the jeopardy criterion used by NMFS in the 1995 FCRPS Biological Opinion is indicated by the dashed horizontal line across each graph (70% for survival criteria; 50% for recovery criteria).

The ability to meet the 24-year survival criterion (**Figure 5.5.1.3-1**) is strongly related to the current status of the stocks, although alternative management actions have some effect on the projected results. In general, the actions involving drawdown of dams result in higher projected probabilities of meeting the 24-year survival criterion. Because the models were not extinction models, this reported ability to meet the survival criterion has to be interpreted with caution, and is likely to be optimistic.

The 48-year projections of performance relative to the recovery criterion (**Figure 5.5.1.3-2**) give the greatest contrast among the alternative hydrosystem actions. Almost all actions involving Snake River drawdown are projected to exceed the 50% recovery performance criteria, on average. In dramatic contrast, A1, A2, and A2' (no drawdown options) fail to meet the recovery criterion in the majority of the runs. In addition, the size of the "middle 50%" box for dam breaching is consistently smaller than the "middle 50%" associated with no breaching options. Thus, breaching is more risk-averse in two ways:

- Breaching consistently yields predicted populations that exceed recovery criteria over a wider range of assumption sets

- The uncertainty (or variability) in outcomes is consistently reduced with breaching (smaller "middle 50%" boxes).

5.5.1.4 The Key Assumptions Underlying Critical Comparisons for Decision Making

The results summarized in **Figures 5.5.1.3-1 and 5.5.1.3-2** display the effects of management actions across all assumption sets, with each assumption weighted as equally likely. One of the strengths of the PATH analytical process is that it allows one to quantify the effects of particular assumptions and thereby identify the most important assumptions. Using a regression tree approach (a technique that quantifies which assumption choices most strongly determine outcomes), PATH reported that the choice of CRiSP versus FLUSH passage models and the source of extra mortality had the greatest influence on results (Marmorek et al. 1998a). To show this graphically, NMFS has focused on the contrast between A1 (current operations, no breaching) and A3 (dams breached in 3 years) and examined how the probability of exceeding recovery criteria depends on these critical assumptions (**Figure 5.5.1.4-1**).

In light of recent PIT-tag data suggesting that D-values may be higher than have been used on average in the PATH simulations (see **Section 5.4**), NMFS ran a series of prospective simulations to examine the effect of higher D-values (and hence lower differential delayed transportation mortality) on the relative probability of meeting the 48-year recovery criterion. The results of these runs, shown in **Figure 5.5.1.4-2**, dramatize the extent to which the performance of management options hinges on the value of D. Using all of the assumption sets, if D = 0.8, the relative reduction in risk would be 11% for dam breaching. This would still represent a substantial reduction in risk (64% average probability of meeting the 48-year recovery criterion versus 53%), but nowhere as dramatic as the 30% reduction in risk associated with the D-values used by PATH. In addition, with a D = 0.8, extra mortality hypotheses become especially important, as shown in **Figure 5.5.1.4-3**. If D= 0.8, breaching may still yield a dramatic reduction in risk (19%), but only if extra mortality is due to the hydrosystem. Indeed, with D = 0.8, if extra mortality is due to an ocean regime shift, then the gains expected with breaching would be negligible (only 2%).

The NMFS is uncertain about the value of D and only further data can resolve that uncertainty. However, the significance of that uncertainty is unarguable. If D-values are low (as has been largely assumed by PATH), breaching would provide a dramatic and compelling reduction in risk across all assumption sets compared to not breaching. However, if D-values are higher (e.g., 0.80 or higher), then the value of breaching depends strongly on what is assumed as the dominant source of extra mortality.

6.0 Fall Chinook Salmon

Unlike spring/summer chinook salmon (which spawn in streams and tributaries), fall chinook salmon are mainstem spawners. Thus, in addition to the effects of the hydrosystem on the survival of juvenile migrants, the hydrosystem directly affects fall chinook salmon by creating reservoirs which submerge and thus eliminate mainstem spawning areas.

As described in the FWCAR (USFWS 1998), the Snake River was considered in some years to be the most important producer of fall chinook salmon in the Columbia River Basin (Fulton 1968). Estimates of fall chinook escapement to spawning areas in the Snake River from 1940 to 1955 averaged 19,447 (range = 3,300 to 30,600) (Irving and Bjornn 1981). Production rates (that is, spawners to returning adults) for Snake River fall chinook salmon from 1940 to 1955 ranged from 1.9:1 to 3.2:1 (Irving and Bjornn 1981). This stock recruitment relationship reflects the healthy status of the Snake River fall chinook salmon population prior to the construction of Hells Canyon complex and the four lower Snake River dams, because the fish were replacing themselves and providing surplus adult production for harvest.

A substantial portion of the historical production of fall chinook salmon in the Snake River originated from areas currently blocked off or inundated by the Hells Canyon complex of dams. Returns to the Snake River system dropped dramatically during the 1960s following completion of the Hells Canyon complex. However, even prior to the Hells Canyon complex of dams, the habitat available to fall chinook salmon had been substantially diminished by the Swan Falls dam in 1901. In recent years, fall chinook salmon spawning in the Snake River may have suffered additional threats because of the presence of significant numbers of hatchery-origin fish (Marmorek et al. 1998a).

6.1 Historical Trends

Direct measures of the annual abundance of individual anadromous fish runs are rarely available. Run-reconstruction techniques were developed to estimate annual escapement and production. Those techniques are generally based upon cohort reconstructions, taking advantage of the information regarding abundance that is available at the time. The following section describes the general approach to reconstructing Columbia River fall chinook salmon runs and provides some details regarding the Deschutes and the Snake River stocks. Reconstructions of additional stocks (Hanford Reach and the North Fork Lewis River runs) were done for comparative purposes and are summarized in Marmorek et al. (1998a).

The Snake River bright fall chinook salmon (SRB) population consists of all adult fall chinook salmon presently spawning in the mainstem Snake River downstream from the Hells Canyon Dam complex to Lower Granite Dam. The existing naturally-spawning fall chinook salmon population is a remnant of a larger run that returned an average of 41,000 spawners annually from 1957 to 1960 (most of which spawned above the Hells Canyon complex of dams). Snake River bright fall chinook salmon migrate a minimum of 720 Rkm past eight mainstem dams on the Snake and Columbia rivers. Approximately 232 Rkm of the mainstem reach above Lower Granite Dam is presently accessible to spawning adults. Habitat quality for spawners and juveniles is considered poor-to-fair relative to habitat used by stocks in the Deschutes River, North Fork Lewis River, and the Columbia River in the Hanford Reach.

Although management actions were evaluated with respect to the Snake River stocks, several additional index stocks were analyzed retrospectively to help distinguish between alternative hypotheses. These comparative populations are described in detail in Marmorek et al. (1998a).

6.1.1 Run Reconstructions

Marmorek et al. (1998a) provides a detailed discussion of the approach to reconstructing fall chinook salmon runs. Annual estimates of escapement are the starting point for the fall chinook salmon run reconstructions. The methods for estimating annual escapements differed among the fall chinook salmon index stocks, reflecting the particular settings and available data. Estimates of the annual number of spawners for each stock are expanded to account for tributary harvest, losses during upstream passage, and mainstem harvest impacts. The resulting estimate represents the annual return to the Columbia River mouth **(Figure 6.1.1-1)**. Each annual return is made up of contributions from several brood years.

6.2 Adult Harvest and Upstream Passage

6.2.1 Harvest Rates

Snake River fall chinook salmon are widely distributed in the ocean and are harvested in fisheries from Alaska to California. Harvest rates in ocean fisheries have generally declined since the early 1980s as a result of restrictions to protect weak or declining stocks in the U.S. and Canada. Ocean-age specific harvest rates are estimated from coded wire tag (CWT) marking experiments. The techniques used reflect the approach employed by the Chinook Technical Committee of the Pacific Salmon Commission for coastwide chinook salmon conservation and rebuilding assessments (CTC 1988). The approach is based upon reconstructing cohorts of CWT-marked fish, incorporating annual estimates of stock specific-ocean harvest based on CWT recoveries and assumptions regarding natural mortality rates during the ocean life-history phase. The result of the CWT cohort analysis is a table of annual estimates of age-specific ocean harvest rates by major fishery. Missing years in the CWT series are filled in using data from adjacent years or through extrapolation from years with CWT data. The natural and hatchery CWT groups available for estimating ocean exploitation rates are shown in **Table 6.2.1-1**.

Snake River fall chinook salmon return at ages 2 through 5, with age-2 returns consisting almost exclusively of males. In some years, returns are predominated by the age-2 component from a particular brood year. Because spawner counts that include 2-year old fish (jacks) do not represent the potential for egg deposition, spawner-recruit analyses rely on returns of 3+-year olds. A summary of annual harvest rates by age class is presented in **Table 6.2.1-1**. Estimates indicate that ocean harvest rates have declined from as high as 50% in the early 1980s to the current level of roughly 20 to 30%.

6.2.2 Upstream Passage

As described in **Section 4.2.2**, estimates of the number of fish lost during upstream migration are based on comparative dam counts recorded by species and general age category (jacks or adults based on length). Annual conversion rates representing non-harvest losses between Bonneville Dam and McNary Dam are calculated for the aggregate up-river bright run, including the Hanford Reach and Snake River populations (**Table 6.2.1-1**). Annual conversion rates are calculated by dividing the adult count at McNary Dam by the count at Bonneville Dam, adjusted to take out estimated escapements (hatchery and tributary) and harvests between the two dams (see formula in **Section 4.2.2)**. The problem with the conversion rates in **Table 6.2.1-1** is that they merely reflect counts of fish at dams. They do not take into account fish that may fall back downstream and never pass a particular dam again, or fish that may fall back and reascend the ladder at a particular dam. This could become problematic if one wants to compare expected increases in adult survival with removal of dams. Where there are well-known fallback problems (e.g., Ice Harbor Dam), this bias is avoided by extrapolating conversion rates from dams with less frequent fallback (the Lower Monumental to Lower Granite Dam segment). For these segments with less of a fallback problem there are some independent survival rates for fall chinook salmon returning to the Snake River that provide a measure of the severity of the bias expected for conversion rates.

Recent analyses of PIT-tagged adults in 1998 (C. Paulsen, Paulson Environmental Research Limited, Lake Oswego, Oregon, Memorandum to PATH life-cycle modelers dated February 17, 1998) allow for an independent estimate of upstream survival to compare against a conversion rate estimate. During collection of fish for radio-tracking studies at Bonneville Dam in 1998, 27 fall chinook salmon adults that were known to have originated above Lower Granite Dam as juveniles were detected at Bonneville Dam. Of these, 10 adults subsequently passed upstream of Lower Granite Dam. The estimated harvest rate in mainstem fisheries on this stock is about 26%. After adjusting for harvest, 50% of the fall chinook salmon (i.e., PIT-tagged adults) passed Lower Granite Dam (i.e., 10/20, where 20 = 0.74 x 27). The estimate for spring/summer chinook salmon is close to the recent average survival rate derived from conversion rate methods. Fall chinook salmon conversion rates show more annual variability. A preliminary PATH conversion rate estimate for 1998 of 0.48 has been calculated for comparison to the conversion rate based on radio-tagging data (C. Paulsen, Paulson Environmental Research Limited, Lake Oswego, Oregon, pers. comm., Memorandum to PATH life-cycle modelers dated February 17, 1998). Consequently, preliminary analyses of the PIT-tag data indicate that the dam-count conversion rates underestimate passage survival by 2% (50% versus 48%).

In general, the effect of fallback at dams is being examined within the PATH process. At this stage, NMFS recognizes there may be some bias built into the estimates of conversion rates but that there are no data to indicate this bias is substantial.

6.3 Egg-to-Smolt Life Stage

Snake River fall chinook salmon spawn in mainstem reaches of the Snake River above Lower Granite Dam and in the lower reaches of major tributaries to the Snake River. After emergence, juvenile fall chinook salmon use mainstem areas for rearing and early growth. Migration to the sea commences in the late spring and early summer of their first year of life.

Recent studies conducted by NMFS and the USFWS (Muir et al. 1998) found that the survival rate of fall chinook salmon marked in mainstem spawning and rearing areas approximately 120 km upstream of Lewiston ranged from 40 to 60% by the time they had migrated to Lower Granite Dam. It took approximately 35 to 55 days for the fish to reach Lower Granite Dam from the time of marking in early May to mid-June. They grew in size from 50 to 70 mm to generally larger than 140 mm when passing Lower Granite Dam. Based on estimates of mortality in reservoirs downstream from Lower Granite Dam, losses in Lower Granite Reservoir and other Snake River reservoirs could be as high as 20%. It is reasonable to assume that drawdown of reservoirs would eliminate much of the high mortality that presently occurs for fall chinook salmon migrants in the lower Snake River.

6.4 Smolt-to-Adult Life Stage

The general timing of the fall chinook salmon outmigration from the Snake River system is known from smolt collections at the mainstem dams. Some information on the relative proportion passing through the different pathways around the dams is available from isolated studies. However, until recently, little direct information existed regarding passage mortality. Beginning with the 1991 outmigration, USFWS initiated a series of PIT-tagging experiments involving Snake River fall chinook salmon releases above the Snake River mainstem dams (Connor et al. 1998). Detections of PIT-tagged fish during their downstream migration have provided detailed information on the characteristics of that migration from 1991 through 1998. That information has provided the basis for adapting the spring/summer chinook salmon passage model for fall chinook salmon. The following section summarizes the key elements of passage survival and the approach used to incorporate those elements into the two passage models. This information is partially a distillation of technical memoranda that were authored by members of the PATH Fall Chinook Hydro/Passage Modeling Work Group. Those documents are archived on a WEB page maintained by University of Washington staff at the Internet address *http://www.cqs.washington.edu/dart/dart.html*. Both "Fall CRiSP" and "Fall FLUSH" use specific flow-rate, reservoir elevation, spill rate, and temperature data in their passage models. These variables influence several mechanisms within 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 salmon 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 simulations are based on historic flow, spill, and elevation data and the prospective simulations are based on output from the hydroregulation models that describe how flows and spills would vary from historical levels under the different flow management scenarios (e.g., A1, the 1995 Biological Opinion, A2, maximize transportation, and A3, drawdown to natural river).

6.4.1 Survival to Below Bonneville Dam

6.4.1.1 Reservoir Survival and Influences of Predation

Loss of subyearling chinook salmon to predators is the primary source of mortality in the reservoirs as simulated in the passage models. Interactions between predators and prey were altered with impoundment of the Columbia and Snake rivers (Bennett and Naughton 1999). Populations of resident predatory fish increased following impoundment by dams (Poe et al. 1991, 1994). In addition, the introduction of non-native species has also greatly changed the composition of the predator assemblage (Li et al. 1987, Poe et al. 1994). Prior to predator introductions (before 1900), northern pikeminnow (previously called northern squawfish), white sturgeon, bull trout, cutthroat trout, and sculpins were probably the major predators in the system. After introductions of non-native species and hydrosystem development, northern pikeminnow, walleye, smallmouth bass, channel catfish, and sculpins are now major predators. The exotic species (bass, walleye, and channel catfish) have undoubtedly increased over the last 100 years, primarily since impoundment (Li et al. 1987), whereas white sturgeon, bull trout, and cutthroat trout are now less abundant. These changes are thought to have occurred because: the extent of slow water habitat preferred by the non-native predators has increased (Poe et al. 1994); dam-induced stress, injury, and disorientation have increased smolt (i.e., prey) 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). In addition, the high level of output of hatchery smolts supports a large predator population that also consumes wild fish.

Rieman et al. (1991) and Beamesderfer and Rieman (1991) observed that the densities and consumption rates of pikeminnows were much higher in the boat restricted zone (BRZ) of the tailrace at John Day Dam than in the John Day Reservoir. However, at Lower Granite Dam, Bennett and Naughton (1999) could detect no difference in pikeminnow predation between these zones.

More importantly, recent PIT-tag studies indicate substantial mortalities for fall chinook migrants in Lower Granite pool (Muir et al. 1998). Survival rates were measured from release points above Lower Granite pool to detection at Lower Granite Dam. In 1997, the survival rate of natural (wild) fish tagged and released early in the season near Pittsburg Landing averaged 57%. The survival to Lower Granite Dam of natural fish released in the vicinity of Billy Creek averaged 32%. Survival rates for hatchery subyearlings released as part of a supplementation program were also low, decreasing through the summer.

Data on predator abundance and consumption rates between 1982 and 1986 is extensive for John Day Reservoir (see Poe and Reiman 1988). A monitoring program has estimated the abundance and consumption for pikeminnow, walleye, smallmouth bass, and catfish relative to John Day Dam 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.

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 will evaluate free-flowing sections in the Snake and Columbia Rivers as well as in reservoir habitat. They will also examine the impact that dams have had on habitat alteration through historic channel mapping. These studies will elucidate how habitat changes from the hydroelectric system may potentially alter predator impacts on juvenile salmonids.

6.4.1.2 Direct Survival at Dams

Juvenile salmonids pass a dam by one of three routes; through turbines, spill, or bypass systems. Several studies have estimated mortality associated with each of these routes of passage and these estimates 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). 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). The fall chinook salmon passage models use a turbine survival estimate of 0.90, which was the same estimate applied to spring/summer chinook salmon in the PATH analyses.

The fall chinook salmon passage workgroup has currently agreed on a value of 0.98 as the survival through the spillway. The ISG (1996) and Whitney et al. (1997) reviewed estimates of spill survival in the Snake and Columbia Rivers published through 1995 and derived a similar survival rate. For initial fall chinook salmon passage model analyses, we adopted 1.0 (the same value used previously in the spring chinook salmon analyses) as the default value for spill effectiveness at all dams except The Dalles Dam.

The mortality of fish that pass a dam via bypass systems was estimated through paired-release experiments at Little Goose Dam conducted by NMFS during 1995 through 1997 (Muir et al. 1998). The experiments conducted in 1995 and 1996 are considered less reliable due to temperature and handling problems. Therefore, the 1997 value only (0.88; S. Smith, Biometrician, NMFS, pers. comm., May 11, 1998) was used for both bypass and sluiceway survival in the current set of passage model analyses. Because of the structure of the experiments (i.e., paired releases), the survival rate reflects the direct mortality that occurs as fish pass through the dam as well as the mortality associated with bypass related predation in the tailrace.

The proportion of juvenile salmonids entering a bypass system is a function of the fish guidance efficiency (FGE) for the different types of screens used to divert the juveniles away from turbines. Two sets of FGEs developed for fall chinook salmon were used in simulations in order to examine model sensitivity to assumptions about the effectiveness of extended length screens (i.e., screens that extend lower into the turbine intake and thus are expected to divert more fish into the bypass system). The first set of FGEs assumed that guidance efficiency remained at the same level reported for standard-length screens while the second set of FGEs assumed an increase in FGEs for extended-length screens. The two sets are described and documented in Marmorek et al. (1998a) and in Krasnow (1997).

A portion of the subyearling chinook salmon collected in bypass collection facilities at Lower Granite, Little Goose, Lower Monumental, and McNary dams are transported. The proportion of fish entering the collection facility is a function of FGE. The transport start and stop dates and the probability of being transported during the collection period determine the proportion of those fish collected that are transported. This information was reported prior to 1982 by NMFS and subsequently by the Corps (**Table 6.4.1.2-1**). The proportion of the fish collected that were transported may not represent the proportion of the migratory population transported because a large fraction of the migratory population may arrive at a collector project after the stop date. Thus, the total proportion of the migratory population that is transported depends not only on the probability of collection at a specific project, but also on the arrival date at that project.

Fish that are transported either through trucks or barges incur some mortality before release below Bonneville Dam. Studies designed to estimate transport survival for subyearling chinook salmon have not been conducted, and hence a value of 0.98 was adopted from the yearling chinook salmon passage model. The value of 0.98, which is used in preliminary analyses, may need to be varied in future simulations to represent uncertainty in direct transportation survival.

6.4.2 Components of Post-Bonneville Dam Mortality

As was the case for spring/summer chinook salmon, the PATH analyses estimated a differential delayed mortality due to transportation for in-river fall chinook salmon. Recall that a D = 1 implies that there is no differential mortality, whereas a low D-value (0.1 – 0.4) reflects a high extra-mortality suffered after transportation. Preliminary simulations to date for fall chinook salmon were run with D-values between 0.14 and 0.32 and thus, high differential mortalities. In addition, it is worth noting that, in 1996, a component of subyearling hatchery releases above Lower Granite Dam were PIT-tagged for passage survival evaluation. Detections at downstream dams during the outmigration combined with adult detections upon return allowed for an estimation of relative survival by passage route. Adult returns through 1998 have been compiled. Given the age at return distribution characteristic of Snake River fall chinook salmon additional returns are expected in 1999 and 2000. The preliminary results are consistent with high differential mortality for transported fall chinook salmon (i.e., a low D). The return rate for fish transported from Lower Granite or Little Goose dam was 0.26%. PIT-tagged fish not passing through bypass/collection systems at dams where fish are transported represent the downriver migrants. Twenty adults returned from an estimated 5,060 juveniles passing through Lower Granite Dam for this group. Assuming that in-river survival from Lower Granite to below Bonneville Dam was 10 to 20% in 1996, the corresponding SAR from below Bonneville to adult return would range from 2 to 4%. The ratio of the transport SAR to these estimates provides an estimated differential survival equivalent to the D-parameter. The ratios for the preliminary returns from 1996 are 0.07 and 0.13, corresponding to assumed in-river survivals of 10 and 20%, respectively. Thus, unlike the case with spring/summer chinook salmon, recent PIT-tag data for fall chinook salmon are not higher than the values used in PATH simulations. In the absence of empirically-derived D-values that are out of line with those used in the PATH simulations, there is less of a need to focus on sensitivity analyses regarding D-values for fall chinook salmon.

Climate factors may also account for changes in survival or recruitment rates. The approach that gave the best fit to historical data was to assume that year-to-year variations in Snake River fall chinook salmon recruitment track changes in the Deschutes River fall chinook salmon stock (Marmorek et al. 1998a). The Deschutes River stock is placed by NMFS in the same Evolutionarily Significant Unit (ESU) as the Snake River stock. Other approaches to estimating a climate factor were much less effective in fitting the time series of spawner-recruit data. These included using temperatures from five Canadian weather stations, indices of year-to-year variations from the spring/summer chinook salmon analysis, and year-to-year variations in the Hanford fall chinook salmon stock.

Post-Bonneville Dam survival of non-transported fish is incorporated into the assessments as a step-decline in survival. Two scenarios were considered in the preliminary assessments. One approach assumes an incremental change in survival taking effect after brood year 1970 (related to the start up of the lower Snake River dams). The other scenario invokes the reduction after brood year 1976 (related to changes in ocean conditions or the full operation of the Snake River dams). The magnitude of this step-wise decline is estimated from the spawner-recruit data. The amount of decline in survival assigned by the model depends not only on the year in which the decline would begin, but also on which passage model is used to estimate system survival. PATH will further develop and explore alternative hypotheses regarding this decline in survival.

Harvest rates represent a final component of post-Bonneville Dam mortality. Ocean harvest impacts on Snake River fall chinook salmon result from management actions throughout a relatively wide range of ocean fisheries. Future changes in those management actions are difficult to predict with certainty. PATH modeled two variations on the retrospective ocean harvest rates as a preliminary sensitivity analysis for the projected results of the different hydrosystem actions, adjusting ocean harvest rates up or down by 15%. This approach reflects the range of harvest rates used to bracket the relationship between catch and abundance proposed for the management of the major northern fisheries in the most recent round of Pacific Salmon Treaty negotiations (Marmorek et al. 1998a). The current Columbia River harvest rules for the fall chinook salmon prospective simulation modeling are based on the 1996-1998 Columbia River Harvest Agreement (1996-1998 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, Oregon). The rules were modified to reflect current run-reconstruction estimates of threshold levels. The schedules are responsive to the status of both the Hanford Reach and Snake River stocks (**Table 6.4.2-1**). During prospective simulations of a drawdown of the Snake River dams, the threshold levels are further modified to reflect improved up-river survival due to the removal of "losses" reflected by low conversion rates.

The preliminary prospective assessments conducted to date have not included sensitivity analyses to alternative Columbia River harvest levels. Sensitivity analyses are currently being designed to explore the projected responses to hydrosystem actions under a range of future ocean/Columbia River harvest scenarios.

6.5 Analysis of Hydrosystem Management Alternatives

The assessment of the potential impacts of alternative management actions involving the lower Snake mainstem dams on fall chinook salmon follows the same general outline as the spring/summer chinook salmon assessment. Briefly, run-reconstruction techniques were employed to create a time series of spawner return estimates bridging the time period when the lower Snake dams were constructed. Alternative assumptions regarding biological mechanisms, climate/environmental effects and the effects of year-by-year actions are then compiled into a retrospective model. A life-cycle modeling approach was used as a framework for analyzing historical trends in the Snake River fall chinook salmon population. In its simplest terms, the fall chinook salmon life-cycle model can be expressed as a basic stock-recruit function modified by factors reflecting juvenile passage survival, climate/ocean effects, and the potential for post-Bonneville Dam survival effects. However, whereas spring/summer chinook salmon assessments considered population parameters for seven index stocks within the Snake River Basin, fall chinook salmon above Lower Granite Dam are treated as a single population. The models were also altered to reflect differences in the life histories of fall and spring/summer chinook salmon. Fall chinook salmon migrate from spawning/early rearing areas in the late spring and summer of their first year of life, whereas Snake River spring/summer chinook salmon migrate in the spring of their second year. Adults return to the Columbia River in late summer and early fall and enter the river intermingled with wild and hatchery runs of fall chinook salmon returning to areas outside of the Snake River Basin. In recent years, the relatively healthy Hanford Reach fall chinook salmon population has dominated the aggregate run of fall chinook salmon returning to the Columbia River.

As for spring/summer chinook salmon, the results of the retrospective analyses were used as the basis for a prospective assessment of the probability that the alternative hydrosystem management actions would result in survival and recovery of Snake River fall chinook salmon. However, the summary provided below is preliminary because PATH has proposed further retrospective and prospective analyses for fall chinook salmon for FY99. Sets of assumptions corresponding to eight factors or uncertainties have been analyzed to date. Each unique combination of the alternative values for the eight factors was simulated using a prospective life-cycle model specifically designed to reflect the information available for Snake River chinook salmon. Details are given in Marmorek et al. (1998a).

6.5.1 Preliminary PATH Results Regarding Management Actions

The fall chinook salmon PATH salmon analyses are recent and have not undergone the same level of regional review as the assessments for spring/summer chinook salmon. Key areas already under examination by the PATH process include: assumptions regarding the implications of PIT-tag results with respect to rearing survival, approaches to estimating potential differential mortality of transported smolts, conversion rates, and the relative performance of different actions under alternative harvest and climate assumptions.

Despite the above caveats about the preliminary nature of the PATH analyses, examination of the results is still informative. Thus far, the prospective modeling indicates that all of the actions analyzed exhibit very high mean probabilities of meeting or exceeding the short-term (24-year) survival escapement levels (**Table 6.5.1-1, Figure 6.5.1-1**). Actions that involve dam breaching also have high mean probabilities of meeting the recovery 48-year recovery escapement levels (**Figure 6.5.1-2, Table 6.5.1-1**). In contrast, transportation actions have substantially lower probabilities of meeting the 48-year recovery escapement levels (**Figure 6.5.1-2**) and fail to meet the recovery criterion of a 50% probability across all assumption sets.

It is noteworthy that, without any detailed PATH modeling, it may be possible to conclude a major benefit to fall chinook salmon as a result of dam breaching. This benefit arises because fall chinook salmon are mainstem spawners and breaching would open up spawning habitat as the reservoirs were drained. Marmorek et al. (1998a) estimate a 77% increase in habitat carrying capacity for fall chinook salmon as a result of breaching. However, this 77% is based simply on an increase in the length of the unimpounded river and does not include subtleties about substrate type (which can dramatically influence the suitability of habitat for fall chinook salmon spawning). Likely habitat improvements for fall chinook salmon are discussed in the FWCAR (USFWS 1998) and in **Section 4.3.3.1**, above.

6.5.2 Quantitative Analysis of Management Options

As noted above, fall chinook salmon production from the Snake River system historically constituted a major portion of the total production of fall chinook salmon from the Columbia River basin (Fulton 1968). The most significant spawning and rearing areas for fall chinook salmon were cut off by the construction of the Hells Canyon complex of dams, upstream from the current mainstem spawning area. The remaining habitat in the Snake River mainstem was further reduced by construction of the four lower Snake River mainstem dams. The Snake River fall chinook salmon population spawning in the mainstem between Hells Canyon Dam and Lower Granite Dam and the lower reaches of major tributaries in that reach along with a population in the Deschutes River are the last remaining population components for this Evolutionarily Significant Unit (Myers et al. 1998). Thus, when discussing the likely effect of lower Snake River drawdown on fall chinook salmon, it is important to put these impacts in the context of the Hells Canyon Dam. In the late 1950s, fall chinook salmon returns to the Snake River system averaged over 40,000 per year and under the best of scenarios, drawdown of dams in the lower Snake River (alternative A3) could not recover even one-quarter of that original amount.

Nonetheless, drawdown of the lower Snake River projects would support the possibility of fall chinook salmon recolonizing historical spawning and rearing areas in the lower Snake River. The lower Snake River dams inundated fall chinook salmon spawning and rearing area that supported up to 5,000 spawners. The reestablishment of a significant fall chinook salmon population lower in the Snake River (i.e., the potential 5,000 fish that might spawn if habitat became available) would increase the probability of maintaining the threatened Snake River ESU as a unique and viable genetic grouping. In addition, drawdown would be likely to ameliorate the high predation losses observed in Lower Granite Reservoir.

In summary, although projected increases in fall chinook salmon due to dam breaching and improved downstream-migrant survival remain preliminary, there is an unquestionable benefit to fall chinook salmon of providing substantially more habitat if option A3 (dam drawdown) is pursued. One does not need a model to conclude that an increase in spawning habitat on the order of 70 to 80% could markedly enhance fall chinook salmon's prospects for survival and recovery. The uncertainty concerns the quality of habitat that would be created if breaching occurred, and how many fish this additional habitat could support. In addition, with breaching, the current high mortality rate of fall chinook salmon in Lower Granite Reservoir would probably be substantially reduced.

7.0 Steelhead

Information on Snake River steelhead (*O. mykiss*) is sketchy because it is nearly impossible to develop stock-specific estimates of abundance and survival. Additionally, it is nearly impossible to obtain accurate redd counts for Snake River steelhead because of their spawning locations and timing. The result of these limitations is a more qualitative than quantitative analysis of effects of proposed actions on this species. Nonetheless, some insight regarding hydrosystem options and the future prospect for survival and recovery of steelhead is possible from comparisons to spring/summer chinook salmon (noting both similarities and contrasts). In particular, to the extent that steelhead respond like spring/summer chinook salmon, we can supplement our limited quantitative data for steelhead with the spring/summer chinook salmon PATH analyses and inferences. There are, of course, limitations to our ability to extrapolate from spring/summer chinook salmon to steelhead.

Biologically, steelhead are divided into two basic run-types, based on the state of sexual maturity at the time of river entry and duration of spawning migration (Burgner et al. 1992). The stream-maturing type, or summer steelhead, enters fresh water in a sexually immature condition and requires several months in freshwater to mature and spawn. The ocean-maturing type, or winter steelhead, enters fresh water with well-developed gonads and spawns shortly after river entry (Barnhart 1986). Snake River steelhead are all classified as summer steelhead. Inland steelhead of the Columbia River Basin, especially the Snake River subbasin, are commonly referred to as either *A-run* or *B-run*. These designations are based on the observation of a bimodal migration of adult steelhead at Bonneville Dam and differences in age (1- versus 2-ocean) and adult size among Snake River steelhead. Adult A-run steelhead enter freshwater from June to August; as defined, the A-run passes Bonneville Dam before 25 August (CBFWA 1990, IDFG 1994). Adult B-run steelhead enter fresh water from late August to October, passing Bonneville Dam after 25 August (CBFWA 1990, IDFG 1994). Above Bonneville Dam, run-timing separation is not observed and the groups are separated based on ocean age and body size (IDFG 1994). A-run steelhead are defined as predominately age-1-ocean, while B-run steelhead are defined as age-2-ocean (IDFG 1994). Adult B-run steelhead are also, on average, 7.5-10 cm larger than A-run steelhead of the same age; this difference is attributed to their longer average residence in salt water (Bjornn 1978, CBFWA 1990, CRFMP TAC 1991). It is unclear, however, if the life history and body size differences observed upstream are correlated with the groups forming the bimodal migration observed at Bonneville Dam. Furthermore, the relationship between patterns observed at the dams and the distribution of adults in spawning areas throughout the Snake River Basin is not well understood.

Steelhead spend between 1 and 4 years in the ocean. Judging from tag returns, most steelhead migrate north and south in the ocean along the continental shelf (Barnhart 1986). Summer steelhead enter fresh water between May and October in the Pacific Northwest (Busby et al. 1996 and Nickelson et al. 1992). They require cool, deep holding pools during summer and fall, prior to spawning (Nickelson et al. 1992). They migrate inland toward spawning areas, overwinter in the larger rivers, resume migrating in early spring to natal streams, and then spawn (Meehan and Bjornn 1991; Nickelson et al. 1992). Steelhead typically spawn between December and June (Bell 1991) and there is a high degree of overlap in timing between populations regardless of run type (Busby et al. 1996). Snow-pack levels at that time of year and the remoteness of spawning grounds contribute to the relative lack of specific information on steelhead spawning. Steelhead eggs generally incubate between February and June (Bell 1991), and typically emerge from the gravel two to three weeks after hatching (Barnhart 1986).

Unlike Pacific salmon, steelhead are capable of spawning multiple times before death. However, it is rare for steelhead to spawn more than twice before dying; most that do so are females (Nickelson et al. 1992). 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 5% (3.4% [Long and Griffin 1937]; 1.6% [Whitt 1954]). The current proportion is unknown, but is assumed near zero.

Steelhead, which spawn in cool, clear streams, arrive at their spawning grounds weeks or even months before they spawn and are vulnerable to disturbance and predation during that period (Barnhart 1986; Everest 1973). Cover, in the form of overhanging vegetation, undercut banks, submerged vegetation, submerged objects such as logs and rocks, floating debris, deep water, turbulence, and turbidity (Giger 1973) are required to reduce disturbance and predation of spawning steelhead. Juvenile steelhead prefer water temperatures ranging from 12 to 15°C (Reeves et al. 1987). They rear in fresh water from 1 to 4 years, then migrate to the ocean as smolts. Steelhead smolts are usually 15 to 20 cm total length and migrate to the ocean in the spring (Meehan and Bjornn 1991).

The Snake River Evolutionarily Significant Unit generally matures after one year in the ocean. Based on data from purse seine catches, juvenile steelhead tend to migrate directly offshore during their first summer from whatever point they enter the ocean rather than migrating along the coastal shelf as do salmon. During fall and winter, juveniles move southward and eastward (Hartt and Dell 1986). Oregon steelhead tend to be north migrating (Nicholas and Hankin 1988; Pearcy et al. 1990, Pearcy 1992).

7.1 Historical Trends

The average return of wild steelhead to the Snake River Basin declined from approximately 30,000 to 80,000 adults in the 1960s through mid-1970s to 7,000 to 30,000 in recent years (**Figure 7.1-1**). Average returns during 1990 through 1991 and for the 1995 and 1996 return years was 11,465 fish. The general pattern has included a sharp decline in abundance in the early 1970s, a modest increasing trend from the mid-1970s through the early 1980s, and another decline during the 1990s. The sharp decline in steelhead numbers during the early 1970s parallels the similar sharp decline in spring/summer chinook salmon populations during the same time period (**Figure 5.1-1**). However, whereas the wild steelhead population in the Snake River doubled from 1975 (13,000) to 1985 (27,000), the spring/summer chinook salmon did not show an increase. In addition, much of the initial steelhead decline in the 1970s may be attributed to the construction of Dworshak Dam in 1973. This dam cut off access to the North Fork of the Clearwater River, which was an important spawning and rearing area for B-run steelhead.

7.2 Adult Harvest and Upstream Passage

7.2.1 Harvest Rates

Snake River steelhead are not targeted by ocean fisheries and ocean harvest of steelhead is effectively non-existent. Columbia River harvest rates have varied as a function of run size (**Figure 7.2.1-1**). When wild Snake River steelhead abundance was relatively high in the 1960s and early 1970s, aggregate (i.e., combined hatchery and wild for all stocks) upriver steelhead harvest rates ranged from 23 to 40% (ODFW and WDFW 1998). As abundance declined through the mid-1970s and partially rebuilt during the early 1980s, aggregate harvest rates dropped, ranging from approximately 6 to 13%. From 1984 through 1993 aggregate harvest rates increased to 16 to 25%, and then dropped again to 10 to 11% since 1994. This description of aggregate harvest rates is representative of mainstem harvest of wild A-run steelhead, but underestimates the wild B-run mainstem harvest rates, which have ranged from approximately 25 to 47% since the mid-1980s (TAC 1997).

In general, trends in harvest rates do not appear to explain trends in abundance. In particular, because both harvest rate and abundance declined in the early 1970s, it is unlikely that harvest rate was a significant cause of that decline. The absence of a negative association between Snake River steelhead harvest rate and abundance does not mean that harvest is unimportant; it simply means that fluctuations in steelhead numbers are not well-explained by fluctuations in harvest rates. It is worth noting that the magnitude of steelhead harvest rates has been, on average, much higher than the magnitude of spring/summer chinook salmon harvest. In particular, since 1991, the wild Snake River spring/summer chinook harvest rate has averaged 5.4% whereas the wild Snake River steelhead harvest rate has averaged 21.6% (Marmorek et al. 1998a).

7.2.2 Upstream Passage

The best estimates of adult steelhead survival through the lower Columbia and lower Snake rivers come from radio-telemetry studies. This method provides an estimate of losses that are not due to harvest, fallbacks, or turn-offs into tributaries. It is generally considered to represent mortality associated with dam passage. A review of radio-telemetry results published to date indicates that average survival of adult steelhead from Bonneville Dam to Lower Granite Dam is approximately 79% (Ross 1998, Marmorek et al. 1998a). This is similar to the estimate of approximately 76% for spring/summer chinook salmon, from the same studies. Translated into a mortality rate, it represents approximately 3% mortality per hydrosystem project.

Are trends in the abundance of Snake River steelhead related to adult passage mortality? Because the number of radio-telemetry studies is limited, it is not possible to make this comparison. A doubling in the number of mainstem dams, from four to eight, between 1968 and 1975 suggests that adult passage mortality could have increased during this period, at least partially explaining the declining trend in abundance. If the current per-project survival of 97% (= $0.79^{1/8}$) occurred prior to 1968, increasing the number of dams from four to eight would have decreased passage survival about 10%, from 89% (= 0.97^{4}) to 79%. However, the greatest decline in spawner returns occurred between 1972 and 1974 (**Figure 7.1-1**), when the number of mainstem dams was constant. In addition, completion of the final dam in 1975 does not appear to be associated with any additional decline in abundance.

Survival of adult Snake River steelhead from the Columbia River mouth to above the site of Lower Granite Dam increased during the late 1960s through early 1970s, when run escapements were trending downward. This increase probably resulted from a decrease in the mainstem harvest rate (from between 23 and 40% to between 6 and 13%) during that period, which most likely outweighed any increase in upstream passage mortality, associated with dam passage. As a result, the decline in Snake River steelhead runs from the late 1960s to the early 1970s is not explained by an increase in adult mortality. The additional decline in the 1990s also cannot be explained by trends in adult mortality, although harvest rates on wild Snake River steelhead, particularly the B-run component, are still comparatively high.

7.3 Egg-to-Smolt Life Stage

The egg to outmigrating smolt stage for Snake River steelhead covers at last three critical time periods – incubation and overwintering in the interstices of the spawning gravels, early rearing in the tributaries, and overwintering as juveniles. It is difficult to follow particular samples of fish through this life stage. Although some information is available for spring/summer chinook salmon, virtually no information exists that is useful for determining trends in steelhead survival during this life stage. Changes in the quantity (particularly loss of habitat in the North Fork Clearwater River) and quality of freshwater spawning and rearing and prespawning habitat may have contributed to production declines in some index streams. However, it is not possible to determine whether there have been recent changes in egg-to-smolt survival. This lack of information also means that we do not know whether the post-1990 decline in Snake River steelhead abundance or the decline in abundance from the late 1960s through the mid-1970s is related to changes in egg-to-smolt survival.

We do know that the declines in returns of Snake River wild steelhead and spring/summer chinook salmon have led to significant decreases in the number of adult carcasses deposited in the natal tributaries. Recent field experiments in western Washington and the Snake basin support the hypothesis that nutrients from adult carcasses contribute to the production of juvenile steelhead (Bilby et al. 1998). Current productivity rates of Snake River steelhead runs may have decreased from historical levels, at least in part, because of the loss in nutrient input from adult carcasses.

7.4 Smolt-to-Adult Life Stage

Survival from the time Snake River steelhead begin their mainstem migration to the ocean until their return as adults (measured as smolt-to-adult returns) accounts for much of the observed decline in run size from the late 1960s through the early 1970s (Marmorek et al. 1998a) (**Figures 7.4-1 and 7.4-2**). The temporal patterning of steelhead SARs also explains much of the population upsurge in the late 1970s, as well as the steelhead population decline in the 1990s.

7.4.1 Direct Survival to Below Bonneville Dam

Mainstem passage survival to below Bonneville Dam can be estimated based on tagging or marking experiments. Methods for estimating mainstem passage survival have developed rapidly in recent years. Estimates for the historical series including the impact of construction and operation of the Snake River dams are based on extrapolations from studies over particular reaches within the system. Until recently, estimates of total direct survival through the entire mainstem from the upper most Snake River project (Lower Granite) to below Bonneville were not possible. The installation of PIT tag detectors at Bonneville Dam, combined with the development of trawl mounted detectors for use in the reach below Bonneville, enabled researchers to develop direct survival estimates over the entire reach during 1997 and 1998 (Smith and Williams 1999).

In contrast to analyses of spring/summer chinook salmon passage described previously, detailed Snake River steelhead passage models have not been developed and reviewed within the PATH process. We can approximate the survival of downstream migrants by examining empirical reach survival estimates and, by making relatively simple assumptions, by expanding average per-project survival to reaches that were not included in the study (Smith and Williams 1999) (**Figure 7.4.1-1**). The expanded estimates in Smith and Williams (1999) for 1994 to 1997 reflect the experience of PIT-tagged downstream migrants (which could go through bypasses at as many as three transport collection projects). These data may overestimate the survival of downstream migrants in the run at large (i.e., by about 10% in the case of spring/summer chinook salmon – for which data exist to quantify the over-estimation of bias, Marmorek et al. 1998a). **Figure 7.4.1-1** has been adjusted for this effect.

The pattern of downstream migrant survival estimates displayed in **Figure 7.4.1-1** suggests that direct survival to below Bonneville Dam declined from the late 1960s through 1970s, which is consistent with the pattern of steelhead adult returns and SARs. However, the pattern of direct downstream migrant survival in recent years is not consistent with the further decline in escapement and SARs observed during the 1990s. The survival rates of steelhead migrating through an eight-dam system during 1995 through 1997 are comparable to the survival rates of mixed wild and hatchery steelhead migrating through 4 to 6 dams during the late 1960s. Because a large proportion of steelhead have been transported since the late 1970s, the total direct survival of combined transported and in-river migrants has been even higher than that indicated in **Figure 7.4.1-1** for recent years.

Synthesizing the above data regarding patterns in direct survival, it appears that direct survival through the hydrosystem does not fully explain the trends in escapement or smolt-to-adult survival for Snake River steelhead. Changes in direct survival through the hydrosystem contributed to the downward trend in SARs that began in the late 1960s and extended through the late 1970s. Low direct survival estimates in the early 1970s are consistent with the downturn in overall survival in the 1970s. The increase in proportion of fish transported and the corresponding increase in direct survival through the late 1970s and 1980s are also consistent with the trend of increasing SARs during this period. However, the second decline in steelhead SAR estimates during the 1990s cannot be explained by direct survival through the hydrosystem. Direct steelhead survival to below Bonneville during that period is estimated to have returned to levels at or above those prevalent prior to the construction of most mainstem Snake River dams. In addition, direct survival of steelhead to below Bonneville Dam appears to be at least as high as that of spring/summer chinook salmon, primarily because efficiency of turbine screens, which guide smolts away from turbines and into bypasses or transport collection facilities, is greater for steelhead than for chinook salmon.

7.4.2 Survival Below Bonneville Dam

To this point, a review of trends in Snake River steelhead adult, adult-to-smolt, and smolt-to-adult survival indicates that the smolt-to-adult life stage survival most closely corresponds to observed trends in abundance (with the possible exception of adult survival as inferred from recent harvest levels). This suggests that the causal factor(s) for observed trends primarily affect the smolt-to-adult life stage. A review of trends in direct survival through the hydrosystem to Bonneville Dam indicates that survival through this portion of the smolt-to-adult life stage corresponds to the SAR pattern from the mid-1960s through the late 1970s. However, because it appears that the most recent decline in Snake River steelhead SAR (in the early 1990s) is unrelated to direct survival through the hydrosystem; the following factors potentially affecting post-Bonneville survival are examined to explain the observed pattern of SARs in the 1990s.

7.4.2.1 Climate Effects As a Factor in Survival Below Bonneville Dam

Coronado-Hernandez (1995) noted strong covariation in survival rate and SAR, as inferred from coded-wire tag returns among 67 steelhead hatchery stocks distributed throughout the Pacific Northwest (Oregon, Washington, Idaho, and British Columbia). An increase in survival from mid-1970s through mid-1980s brood years and a decline beginning with late 1980s brood years were particularly evident for summer steelhead (Coronado-Hernandez 1995). When 2 to 3 years are added to each brood year to represent the out-migration year, this pattern matches that of wild Snake River steelhead SARs during the same period. Coronado-Hernandez concluded that a change in ocean climate conditions is the most likely explanation for this type of correspondence among a large number of hatchery stocks. Cooper and Johnson 1992 compared trends among wild and hatchery steelhead stocks from diverse locations along the Pacific coast and reached the same conclusion.

Welch et al. (in press) described a sharp decline in SARs for Keogh River (British Columbia) steelhead during the 1990 through 1994 ocean-entry years, compared to SARs during the 1977 through 1989 period. Trends prior to 1990 were associated with size of smolts at time of ocean entry, but this association was not observed in subsequent years. The authors suggested that the trend in declining SARs is associated with anomalous atmospheric conditions that began in 1989 (Watanabe and Nitta 1999), resulting in a general warming of the central North Pacific after 1977 and anomalous ocean conditions throughout much of the Northeast Pacific after 1990. Based on the condition (i.e., size) of sockeye salmon returning to British Columbia, the authors suggested that the anomalous ocean conditions have affected salmonid growth and survival, although they did not identify specific oceanographic mechanisms. Mantua et al.’s (1997) Pacific Decadal Oscillation Index (PDO) was strongly negative in the early 1990s and has fluctuated during the mid-1990s. The index was mostly positive from about 1978 through 1989 and mostly negative from 1948 through1977 (**Figure 4.4.1-4**, upper graph). This suggests that a more recent shift in climate could at least partially account for the second decline in steelhead SARs since 1990.

7.4.2.2 Indirect Mortality Due to Hydrosystem Passage

A second possible factor influencing post-Bonneville Dam survival is mortality caused by passage experiences above the dam, which are then expressed below Bonneville. Indirect survival effects caused by passage through the hydrosystem could fall into two areas:

- Reductions in the survival of transported fish from release to returns, relative to that of non-transported fish and

- General delayed impacts on both transported and non-transported fish, taking effect below Bonneville Dam.

A preliminary analysis of the relative post-Bonneville Dam survival of transported steelhead, compared to steelhead that were not transported, has been conducted using methods identical to those described for spring/summer chinook salmon in **Section 5.4** of this report. The relative post-Bonneville survival of hatchery steelhead in 1995 (approximately 0.32) is considerably lower than that of hatchery spring/summer chinook salmon during that year (approximately 0.87, Smith and Williams 1999). No other comparisons are available at this time.

General Delayed Impacts on Both Transported and Non-Transported Fish

Sandford and Smith (manuscript submitted for publication) describe recent PIT-tag returns that indicate the SARs of steelhead smolts vary with route of passage through the hydrosystem. This suggests that post-Bonneville mortality is not equivalent for all fish migrating in-river and that the experience of a smolt passing through the hydrosystem, in part, determines the likelihood of survival. Possible mechanisms for this delayed mortality of both transported and non-transported fish, as a result of hydrosystem passage, have been proposed and are described in Marmorek et al. (1998a).

7.4.3 Reduced Stock Viability and Extra Mortality Caused by Factors Other than Hydrosystem Passage

As was the case with spring/summer chinook salmon, there are several alternative hypotheses that explain the extra mortality in of Snake River steelhead. The “reduced stock viability” hypothesis proposes that the viability of Snake River stocks declined since the early 1970s. Under this set of assumptions, at least a portion of the extra mortality is not directly related to either the hydrosystem or to climate conditions. The original mechanism for decreased stock viability was that hatchery programs implemented after construction of the Snake River dams increased either the incidence or the severity of bacterial kidney disease (BKD) within the wild population. As a result, it was hypothesized that mortality increased in juvenile fish after they exited the hydrosystem as compared to years before construction of the Snake River dams. An alternative mechanism has been proposed involving stress due to interactions of migrating wild Snake River chinook salmon with large numbers of hatchery fish released in the system. Evidence from laboratory and field studies supports the assumption that interactions with hatchery fish, in particular large steelhead smolts, can lead to increased stress in spring/summer chinook salmon smolts. This hypothesis is less likely to be true for steelhead than for spring/summer chinook salmon because the pattern of increasing returns during the late 1970s and 1980s is not consistent with the pattern of increasing hatchery releases during the same period (**Figure 7.1-1**). However, it is possible that negative effects of hatchery fish on wild steelhead survival may not match the temporal pattern of hatchery releases (e.g., we might expect a lag in genetic consequences or in ecological interactions that are mediated through changes in habitat quality).

7.5 Examining Alternative Management Actions

The potential effects on steelhead of implementing alternative actions to address Snake River hydrosystem impacts were not analyzed through the PATH process in the same manner as the effects on spring/summer chinook salmon. Rather, conclusions regarding steelhead were derived by inference from the spring/summer chinook salmon analysis as follows:

1. Determine whether spring/summer chinook salmon management actions that result in an acceptable probability exceeding survival threshold population 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 population levels and reaching recovery levels.

3. Define the incremental change from recent steelhead SARs that is necessary to achieve historical SARs.

4. Compare the incremental change in steelhead survival 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 salmon 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 salmon survival outside the hydrosystem.

7. Assume that <u>if</u>:

    (a) spring/summer chinook salmon management actions that result in an acceptable probability of exceeding survival threshold population levels and reaching recovery levels correspond to historical smolt-to-adult survival rates (SAR);

    <u>then</u> historical SARs are a reasonable proxy for an acceptable probability of survival and recovery in Snake River spring/summer chinook salmon <u>and</u> that this approach extends to Snake River steelhead.

    Assume further that <u>if</u>:

    (b) the incremental change between current and historical SAR is less than or equal to the incremental change for spring/summer chinook salmon;

    (c) the management action is likely to have a similar effect on both Snake River steelhead and spring/summer chinook salmon direct <u>hydrosystem survival</u>;

(d) the management action is likely to have a similar effect on both Snake River steelhead and spring/summer chinook salmon survival outside of the hydrosystem;

(e) a management action results in an acceptable probability of Snake River spring/summer chinook salmon 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 one adopts the logic embodied in the above seven-step process, then in conjunction with what we know directly about steelhead, we can draw some conclusions. The major conclusions are:

1. NMFS agrees with the PATH conclusion that actions that resulted in an acceptable probability of meeting the 100-year survival threshold and the 48-year recovery goal were associated with estimated SARs that were within the range of historical SARs (**Figure 7.5-1**). To ensure that populations remain above survival thresholds over the next 24 years, escapement SARs may be required that are somewhat higher than those observed during the historical period.

2. NMFS agrees with PATH that, based on the information presented in **Table 7.5-1** and **7.5-2**, the incremental change between current and historical SAR is less than or equal to the incremental change for spring/summer chinook salmon. Choice of "historical period" for Snake River steelhead is subject to judgement and choice of alternative years and could influence the necessary incremental change. However, even with certain alternative time periods for which historical estimates exist, which were discussed by the PATH steelhead work group, this conclusion would not change. Similarly, the conclusion is not affected by choice of a SAR standard (escapement to upper dam versus escapement plus harvest).

3. NMFS agrees with PATH that, based on an extensive comparison of steelhead and chinook salmon routing and survival through the hydrosystem (Marmorek et al. 1998a), management actions are likely to have similar effects on the direct hydrosystem survival of Snake River steelhead and spring/summer chinook salmon.

4. Although NMFS agrees with PATH that the response of steelhead survival outside the hydrosystem is likely to be similar to that of spring/summer chinook salmon, reservations are warranted because of the poor correspondence in SARs between the species during the mid- to late-1980s, when steelhead SARs were equivalent to those observed in the 1960s, but spring/summer chinook salmon SARs declined to much lower levels. 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 salmon.

5. Actions that meet jeopardy criteria for spring/summer chinook salmon would likely satisfy the biological requirements necessary for survival and recovery of steelhead. This is because steelhead will not require as great a boost in SARs to achieve the needed increase in population levels. However, we cannot rule out the possibility that actions that would fail to

meet survival and recovery criteria for spring/summer chinook salmon would succeed for steelhead.

8.0 Sockeye Salmon

Snake River sockeye salmon are the most depleted of the anadromous fish considered in this report. These stocks constitute an ESU and have been declared as endangered under the ESA. There are so few fish from this ESU in the river that it is impossible to experimentally measure the impact of the hydrosystem on their passage survival. This situation is not likely to change because the number of sockeye salmon that can be outplanted from the captive broodstock program is limited by the carrying capacity of the accessible spawning lakes in the Stanley River Basin. Since 1991, all fish returning to Redfish Lake, the last of the natural spawning areas, have been sequestered in a captive broodstock program to allow the population to persist and to allow reseeding of natural areas. This narrative describes the status of the Snake River ESU over time, conservation efforts (through a captive broodstock program), and the apparent effects of environmental factors in the adult, egg-to-smolt, and smolt-to-adult return life stages.

8.1 Historical Trends

The life history of the sockeye salmon (*O. nerka*) is perhaps the most complex of any Pacific salmon. Multiple forms of the species are common. The species most commonly exhibits two life-history types: an anadromous form (called sockeye salmon) and a nonanadromous (resident) freshwater form (called kokanee). Kokanee progeny occasionally migrate to the sea and return as adults but there is only scattered evidence that these fish contribute to any sockeye salmon population. Kokanee in the Snake River Basin are not considered part of the listed Evolutionarily Significant Unit. A third form, known as residual sockeye salmon (or “residuals”), often occurs together with anadromous sockeye salmon. Residuals are thought to be the progeny of (or recent descendents from anadromous sockeye salmon) but are generally nonanadromous themselves. Wild residuals in the Snake River Basin are part of the listed Evolutionarily Significant Unit.

Historically, Snake River sockeye salmon were produced in the Stanley River Subbasin of Idaho’s Salmon River in Alturas, Pettit, Redfish, and Stanley Lakes and in Warm Lake on the south fork of the South Fork salmon. Sockeye salmon 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 1894, Toner 1960, Bjornn et al. 1968, Fulton 1970).

The largest single sockeye salmon spawning area was in the headwaters of the Payette River, where 75,000 were taken one year by a single fishing operation in Big Payette Lake. However, access to production areas in the Payette Basin was eliminated by construction of Black Canyon Dam in 1924. During the 1880s, returns to headwaters of the Grand Ronde River in Oregon (Wallowa Lake) were estimated to have been at least 24,000 and 30,000 sockeye salmon (Cramer 1990), but access to the Grande Ronde was eliminated by construction of a dam on the outlet to Wallowa Lake in 1929. Access to spawning areas in the upper Snake River Basin was eliminated in 1967 when fish were no longer trapped and transported around the Hells Canyon dam complex. All of these dams were constructed without fish passage facilities.

There are no reliable estimates of the number of sockeye salmon spawning in Redfish Lake at the turn of the century. However, beginning in 1910, access to all lakes in the Stanley Basin was seriously reduced by the construction of Sunbeam Dam, 20 miles downstream 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). It was replaced with a concrete structure in 1920 but sockeye salmon access was impeded until the dam was partially removed in 1934.

Even after fish passage was restored at Sunbeam Dam, sockeye salmon were unable to use spawning areas in two of the lakes in the Stanley Basin. Welsh (1991) reported fish eradication projects in Pettit Lake (treated with toxaphene in 1960) and Stanley Lake (treated with Fish-Tox, a mixture of rotenone and toxaphene, in 1954). Agricultural water diversions cut off access to most of the lakes, as discussed in **Section 8.2.2.4**. Bjornn et al. (1968) stated that, during the 1950s and 1960s, Redfish Lake was probably the only lake in Idaho that was still used by sockeye salmon each year for spawning and rearing and, at the time of listing under the Endangered Species Act (November 20, 1991; FR 56 No. 224), sockeye salmon were produced naturally only in Redfish Lake.

Escapement to the Snake River has declined dramatically in recent years. Adult counts at Ice Harbor Dam have fallen from 3,170 in 1965 to zero in 1990 (**Figure 8.1-1;** ODFW and WDFW 1998). The Idaho Department of Fish and Game counted adults at a weir in Redfish Lake Creek during 1954 through 1966. Adult counts dropped from 4,361 in 1955 to fewer than 500 after 1957 (Bjornn et al. 1968). Fewer than 20 wild adult sockeye salmon returned to Redfish Lake in recent years (1991 through 1998; C. Petrosky, pers. comm., Fishery Biologist, IDFG, December 1, 1998)**.**

8.2 Adult Harvest and Upstream Passage

8.2.1 Harvest

Although historical mainstem harvest rates on Snake River sockeye salmon have been variable, they were generally higher before than after the completion of the hydrosystem (**Figure 8.2.1-1**). Annual mainstem harvest averaged 40% of adults that returned to the Columbia River mouth (range = 0 to 86%) before 1974 and 9% (range = 0 to 49%) after that time (ODFW and WDFW 1998). Thus, the level of harvest on adult returns declined as the effect of hydrosystem passage on juvenile and adult migrants was increasing. No commercial harvest of sockeye salmon has been allowed since 1988, other than a minor incidental catch during the tribal fall-season commercial chinook salmon and steelhead fisheries (ODFW and WDFW 1995). Sockeye salmon fisheries are now managed according to the 1996-1998 Management Agreement, which allows impacts on sockeye salmon of no more than one percent in the non-Indian commercial and recreational fisheries combined.

8.2.2 Upstream Passage

Peak passage of sockeye salmon at Bonneville Dam has occurred during June in recent years. Snake River sockeye salmon (probably the adult progeny of wild residual matings) pass Lower Granite Dam from June 25 to August 30 (USFWS 1998).

8.2.2.1 Per-Project Mortality Rates

Redfish Lake spawner counts declined steeply from 1955 through 1966, the period during which the number of hydroelectric projects on the mainstem doubled from three to six (**Figure 8.2.2.1-1**). Although development of the mainstem hydrosystem coincided in time with other factors affecting the survival of Snake River sockeye salmon, it is reasonable to consider hydrosystem sources of adult loss during migration.

Using conversion rate calculations based on dam counts (**Section 5.2.2**), Ross (1995) estimated a 15.4% rate of loss of adult sockeye salmon between Bonneville and Lower Granite dams. Given the low spawning escapement of Snake River wild sockeye salmon during recent years (**Section 8.1**), the dam counts, and therefore conversion rate estimates for this species probably include wild residuals and anadromous kokanee.

In 1997, researchers from NMFS and the University of Idaho (UI) implanted radiotags in approximately 800 adult sockeye salmon at Bonneville Dam and monitored their upstream migration. A preliminary analysis of the detection records indicated a loss of 11%:

$$(1 - 0.11)^{1/4} = 0.97 \quad \text{[97\% per-project survival]}$$

$$(1 - 0.97) = 0.03 \quad \text{[3\% per-project mortality]}$$

over the four-dam reach between Bonneville and McNary Dams (L. Stuehrenberg, NMFS Northwest Fisheries Science Center, pers. comm., December 17, 1998). All of the tagged fish that were detected by the radio receivers returned to the mid-Columbia reach (i.e., Wenatchee and Okanogan stocks). The single fish that turned off into the Snake River was detected as a fallback at Ice Harbor Dam.

If the following two assumptions are valid:

- The per-dam rate of loss of adult Snake River sockeye salmon in the lower Columbia River is similar to that of individuals from the mid-Columbia stocks and

- The per-dam rate of loss of adult Snake River sockeye salmon through the lower Snake reach would be similar to that measured for mid-Columbia sockeye salmon in the lower Columbia reach,

then data from the 1997 radiotelemetry study indicate a 22% loss through the eight-dam hydrosystem between Bonneville and Lower Granite dams:

$$(0.97)^8 = 0.78 \quad \text{[78\% system survival]}$$

$$(1 - 0.78) = 0.22 \quad \text{[22\% system mortality]}$$

We cannot test the first assumption because radio-telemetry experiments would require more wild adult Snake River sockeye salmon than are in the system. Data from Bjornn et al. (1995) for spring/summer chinook salmon and steelhead indicate that the second assumption would probably result in a slight overestimate of survival through the eight-project Federal hydrosystem (because survival appears to be slightly lower in the lower Snake River; **Table 8.2.2.1-1**).

This calculation of 78% survival for adult Snake River sockeye salmon passing through the eight hydro projects (Bonneville to Lower Granite dams) is similar to 76% survival for Snake River spring/summer chinook salmon and 79% survival for summer steelhead over the same eight-project reach (C. Ross, Fishery Biologist, NMFS, pers. comm., February 23, 1999).

8.2.2.2 Migration Rates

No data are available on the migration rates of adult sockeye salmon through the lower Snake River or the free-flowing reach above Lower Granite Reservoir. Quinn et al. (1997) compared travel rates (days between 50% passage dates) for adult sockeye salmon between Bonneville and McNary dams to flow (mean daily discharge during June and July) for the period 1954 to 1994. Travel rate was negatively correlated with flow at McNary Dam; fish traveled faster as flow decreased. Warmer water at McNary Dam was also associated with faster travel rates. Although not specified by Quinn et al., these fish are likely to be a mixture of sockeye salmon from the Snake River and upper-Columbia River ESU, wild residual sockeye salmon from both ESUs, and anadromous kokanee from upstream storage reservoirs in the Snake and Columbia River systems.

8.2.2.3 Access to Spawning Grounds

At this time, anadromous fish passage remains cut off to all former Snake River sockeye salmon habitat except that in the Stanley Basin. Chapman et al. (1990) cite agricultural diversions as a cause of the sockeye salmon's decline from all Stanley Basin lakes, including Redfish Lake. They note that more than 68 agricultural diversions are present on the Salmon River and tributaries within the Sawtooth National Recreation Area. The diversion at Busterback Ranch, on Alturas Lake Creek in the Stanley Basin, dewatered the creek, completely blocking sockeye salmon from Alturas Lake (Bowles and Cochnaeur 1984, Chapman et al. 1990, IDFG 1998). Although some diversions in the Salmon River Basin have been screened since the mid-1950s (Delarm and Wold 1985), many of those diversions in Stanley River Subbasin streams were not screened until the mid- to late 1970s and some are still not screened.

At the present time, an aggressive screen replacement and construction program, funded through the Mitchell Act, is improving conditions on the mainstem Salmon River for juvenile sockeye salmon. Activities include the installation of state-of-the-art fish screens and bypass return systems. Busterback Ranch no longer diverts instream flows because the U.S. Forest Service (USFS), using BPA funds, purchased the water right. In addition, the U.S. Bureau of Reclamation (USBR) has been actively correcting problems at agricultural diversions on the mainstem Salmon River.

Dewatering of streams is an ongoing habitat problem. Idaho water law allows the diversion of flows in excess of water rights as long as downstream water rights are not affected. In addition, water rights for fish-screen bypass returns are junior to agricultural water rights, allowing a water user to shut off the fish bypass when the senior water right cannot be diverted.

Overall, sockeye salmon, which rear in lakes, may be less vulnerable to the negative effects of agricultural practices than spring/summer chinook salmon, which rear in streams. Water quality in Redfish Lake is very good and an adequate amount of spawning habitat is available (T. Flagg, NMFS representative to the SBTOC, pers. comm., January 6, 1999). However, future improvements to spawning habitat conditions must be treated as an uncertainty in any evaluation of the probability that an alternative hydrosystem action would result in survival and recovery of Snake River sockeye salmon.

8.2.2.4 Spawning Population Size

Spawning ground surveys in Redfish Lake during 1988 identified four adults and two redds. One adult sockeye salmon, one redd, and a second potential redd were observed during 1989. No redds or adults were observed during 1990. Since 1991, all adult sockeye salmon returning to Redfish Lake have been trapped at the weir and taken into the captive broodstock program (Pravecek and Johnson 1997; Kline and Lamansky 1997). An emergency artificial propagation (captive broodstock) program was begun in 1991 to preserve Redfish Lake sockeye salmon, believed to be the only remaining stock in the Snake River basin. The broodstock program is administered by NMFS, the Idaho Department of Fish and Game (IDFG), the Shoshone-Bannock Tribe (SBT), the University of Idaho (UI), the Idaho Department of Environmental Quality (IDEQ), and the BPA through the Stanley Basin Technical Oversight Committee (SBTOC). In contrast to a traditional hatchery program, which outplants smolts each year, sockeye salmon are cultured in captivity for a complete life cycle.

The progeny of captively reared adults are then released to supplement wild populations. The purpose of the program is to maintain the species and prevent extinction in the short term and to "jump start" the reestablishment of sockeye salmon runs to the waters of the Stanley basin in the long term. Ultimately, regional fish and wildlife managers hope to rebuild stocks to levels that will allow consumptive use of Snake River sockeye salmon and kokanee (IDFG 1998). Approximately 40 redds were counted after IDFG released 120 adults into Redfish Lake in September 1996 (IDFG 1998). In 1997, when researchers released 80 adults, they counted about 30 redds in Redfish Lake, one redd in Pettit Lake, and some test digs in Alturas Lake. The long-term success of these fish in producing offspring and adult returns is, as yet, unknown.

8.3 Egg-to-Smolt Stage

During 1998, NMFS and IDFG released approximately 160,000 subyearling parr (presmolts) and smolts from the sockeye salmon captive broodstock program to Stanley Basin lakes. These releases were comprised of second-generation progeny from the 1993 and 1994 brood years and third-generation progeny from the 1991 brood year. As previously stated, despite ongoing outplants of hatchery fish, the regional fish and wildlife managers do not expect the captive broodstock program, by itself, to produce self-sustaining, naturally reproducing populations of Snake River sockeye salmon. Despite efforts by the SBTOC to increase the carrying capacity of the available spawning lakes, the limited number of spawning lakes with unimpeded passage to the mainstem continues to limit the number of sockeye salmon presmolts that can be outplanted to overwinter in the wild. Thus, although it may be possible to achieve the recovery of the Stanley River Basin population, the number of wild sockeye salmon in the system will remain at numbers below those needed to support quantitative research regarding the effects of passage through the hydrosystem.

8.4 Smolt-to-Adult Return Stage

The number of hydroelectric projects on the mainstem doubled from three to six during the period 1960 through 1969. Smolt-to-adult return rates (SAR) for 1955 through 1964 averaged 0.8% (Bjornn et al. 1968). During 1991 through 1996, average SAR declined by over 90% to 0.07% (C Petrosky, Fishery Biologist, IDFG, pers. comm., December 21, 1998) (**Figure 8.4-1**). These SARs represent the survival rates of wild residual smolts from Redfish Lake which have returned as adults ("escapement SAR" as defined in NMFS 1998).

As with other Snake River salmonids, the decline of Snake River sockeye salmon corresponds in time with other trends besides development of the hydrosystem. These include the addition of unscreened diversion in tributaries connecting spawning areas with the mainstem and construction of dams that blocked fish passage (**Section 8.2.2.3**). Beginning in the late 1970s, ocean environmental conditions changed, as did the quantity of hatchery salmonid production. Mechanisms associated with these coincidental trends have been hypothesized as alternative or at least contributory explanatory variables for the decline of other Snake River salmonids.

8.4.1 Survival of Juvenile Sockeye Salmon Through the Hydrosystem

Juvenile sockeye salmon typically outmigrate over an extended period. Earlier reports indicated that sockeye salmon smolts left nursery areas in the Snake River Subbasin during May and June. Recent index counts show that wild sockeye salmon pass Lower Granite Dam from March through early September with the outmigration continuing into November (data compiled by Fish Passage Center; reported in USFWS 1998). In comparison, the index counts for Rock Island Dam on the mid-Columbia River show sockeye salmon passage during mid-April through mid-July (USFWS 1998). The more protracted outmigration in the lower Snake River may reflect differences in the run timing of wild residuals or of kokanee washing out of upstream reservoirs.

The limited data describing fish guidance efficiencies (FGE) for sockeye salmon at mainstem dams indicate that, where standard-length submersible screens (STS) are used, FGEs may be somewhat lower than those observed for spring/summer chinook salmon. Although sockeye salmon guidance increased where standard-length screens were lowered farther into the turbine intake, it was still lower than that of spring/summer chinook salmon. Only where extended-length bar screens (ESBS) were used did sockeye salmon guidance rise to that of spring/summer chinook salmon (**Table 8.4.1-3**).

Descaling rates for sockeye salmon at lower Snake River and McNary dams may indicate a mechanism for increased mortality resulting from dam passage. Descaling rates for the period 1981 through 1997 are shown in **Table 8.4.1-2**.

These data, when compared with similar estimates for steelhead and spring/summer chinook salmon (Marmorek et al. 1998a), indicate that descaling rates are substantially higher for hatchery and wild residual sockeye salmon/wild anadromous kokanee than for other salmonids for which data are available (Marmorek et al. 1998a). 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 salmon/wild residuals/anadromous kokanee appear to have experienced greater descaling rates than hatchery sockeye salmon. Although no estimates of reach-survival are available for juvenile sockeye salmon, it is reasonable to assume that descaling causes the reach survival of sockeye salmon to be lower than that observed for other Snake River salmonids.

Neither the direct nor indirect transport survival of Snake River sockeye salmon has been evaluated. No information is available regarding the relative SARs of transported and non-transported fish. Transport-survival studies for sockeye salmon trucked from Priest Rapids Dam were performed during 1984 through 1988. However, Chapman et al. (1997) reviewed these studies and concluded that the protocols were specific to the mid-Columbia reach and these data should not be used in comparative evaluations of transport-survival from the lower Snake River or McNary Dam.

Predation studies have not been conducted with juvenile Snake River sockeye salmon migrating through either the mainstem Snake or Columbia River. Zimmerman (1997) reported that approximately 85% of the identifiable fish in the guts of northern pikeminnow from lower Snake River reservoirs were salmonids. Of these, 50% could not be identified to species. Even if some prey items had been identified as sockeye salmon, without tags, researchers would not be able to determine whether the sockeye salmon originated from the stocks in the Clearwater or Stanley Subbasin. Thus, predation on juvenile sockeye salmon in mainstem reservoirs must be treated as an uncertainty in any evaluation of the probability that an alternative hydrosystem action would result in the survival and recovery of Snake River sockeye salmon.

8.5 Effects of Ocean and Estuarine Conditions

As described in **Section 4.4.2**, survival through the estuary and ocean life-history phase is affected by year-to-year variation and multi-year trends in climate and environmental effects. There are no available data on the oceanic distribution of Snake River sockeye salmon or wild residuals from the Evolutionarily Significant Unit. Therefore, it is not possible to predict the degree to which changes in ocean conditions have influenced the decline of this Evolutionarily Significant Unit or will contribute to its recovery.

Fryer (1998) reported that the percentages of both sockeye salmon and spring/summer chinook salmon passing Bonneville Dam with pinniped-caused abrasions increased between 1991 and 1996. However, he noted that these trends could not be used to determine whether pinniped predation was a significant source of mortality during that period.

No data are available on rates of predation on juvenile sockeye salmon by fish-eating birds. Because relatively few juvenile Snake River sockeye salmon are tagged, recoveries at bird colonies are expected to be low. However, the potential exists for significant predation on those outplants from the captive broodstock program that survive passage through the hydrosystem. This factor must be treated as an uncertainty in any evaluation of the probability that alternative hydrosystem actions would result in survival and recovery of Snake River sockeye salmon.

8.6 Effects of Hatchery Releases

Williams et al. (1998a) hypothesized that hatchery releases (especially extensive releases of large steelhead smolts) contributed to extra (post-Bonneville) mortality in spring/summer chinook salmon by reducing growth rate and increasing stress, predation, and disease transmission. These negative effects may also apply to sockeye salmon, albeit to an unknown degree. In contrast, the potential effects of hatchery programs on the genetic integrity of the Snake River sockeye salmon Evolutionarily Significant Unit (i.e., increase in demographic and catastrophic risks of extinction, loss of genetic diversity within and among populations, and domestication) are not a significant concern, at least at the present time. The only Snake River hatchery program is the emergency captive broodstock for Redfish Lake; although this program entails genetic and other risks to this Evolutionarily Significant Unit, these risks are considered to be less than the risk of not intervening. Whereas hatchery production of spring/summer chinook salmon was conceived as a means to augment harvest and began as early as the late 19th century (SRT and ISAB 1998), the captive broodstock program was conceived and developed at the time of listing (1991), with the only alternative nearly certain extirpation. These same concerns could eventually apply to the sockeye salmon hatchery program in the long run if efforts to restore naturally reproducing populations were prolonged. For that reason, the Stanley Basin Technical Oversight Committee is not likely to continue the captive broodstock program indefinitely if ongoing sources of mortality elsewhere in the life cycle are not reversed.

8.7 Relevance to the Analysis of Hydrosystem Management Alternatives

We do not have the option of waiting for further research on the passage survival of Snake River sockeye salmon. The carrying capacity of the Stanley Basin limits the number of fish that can be outplanted to numbers below those needed for quantitative field studies that would resolve questions such as:

- What are the survival rates to Lower Granite Dam of smolts from both the captive broodstock program and from wild residual matings?
- How do environmental conditions affect SARs for both groups?
- What are reach survivals in the lower Snake and lower Columbia rivers for both groups?
- What are the guidance efficiencies at mainstem hydropower projects (especially Lower Granite Dam) for both groups?
- What are the relative smolt-to-adult survival rates for transported fish and in-river migrants for both groups (and how do these vary with in-river conditions and in-river migration routes)?

Because the various life-history forms are not distinguished in the existing literature, we cannot even be sure whether the available data reflect observations of wild sockeye salmon or wild residuals versus anadromous kokanee (the latter are not part of the ESU). It is therefore not possible to consider the likely effects of hydrosystem management options by reference to the prospective analyses for spring/summer (or fall) chinook salmon, as was done for steelhead in **Section 7.0**. However, it is reasonable to assume that the hydrosystem management options that improve opportunities for survival and recovery of chinook salmon will also improve those opportunities for sockeye salmon. But, there are no data to go beyond this generic “plausibility” argument.

9.0 Summary of Management Option Analyses

9.1 Overview of Results for Each Species

Based on the PATH analyses conducted to date, actions involving drawdown (A3 and B1) projected consistently higher relative probabilities of exceeding survival and recovery population thresholds than actions using the current configuration of the hydrosystem (A1 and A2). In general, differences in relative probabilities were most pronounced for the 48-year recovery population thresholds. The degree of difference within analyses for each species varied considerably as a function of the key assumptions incorporated.

9.1.1 Spring/Summer Chinook Salmon

The PATH results indicate that drawdown meets survival and recovery criteria for spring/summer chinook salmon over the widest range of assumption sets. However, simulation runs made by NMFS with lower differential delayed transportation mortality indicate all conclusions about hydrosystem management options for spring/summer chinook salmon are strikingly dependent on assumptions regarding transportation mortality. If transportation can be made as successful as recent PIT-tag data suggest, then the advantages of dam breaching are not as compelling across all sets of assumptions. In particular, if transportation mortality is low, then identifying other causes of “extra mortality” becomes extremely important. Whereas there are advantages to delaying and learning more about these uncertainties regarding transportation and extra mortality, there are also substantial risks involved with delay. Some of the spring/summer chinook salmon index stocks are at very low numbers (**Figure 5.1-1**) and hence in peril of extinction over the short term; moreover, even without considering the imminent risk of extinction, PATH analyses indicate roughly an 8% reduction in the mean probability of meeting survival threshold population levels over the next 24 years if breaching is not pursued.

9.1.2 Fall Chinook Salmon

Preliminary PATH life-cycle modeling results indicate that survival and recovery population thresholds for Snake River fall chinook salmon are more likely to be met with dam breaching than with the current configuration (A2) or with the addition of major system improvements (A2’). However, PATH analyses for fall chinook salmon are still ongoing and quantitative comments about relative probabilities (between hydrosystem options) are premature.

Importantly, fall chinook salmon are mainstem spawners. This means that some improvements due to dam breaching can be identified without detailed modeling. In particular, breaching of the four lower Snake River dams provides not only potential passage survival improvements, but also increases in carrying capacity. The preliminary PATH analyses suggest an increase of 77% in fall chinook salmon carrying capacity under dam breaching (measured in terms of an increase in the length of the unimpounded river). The reopening of fall chinook salmon habitat inundated by the existing lower Snake River dams could serve as a positive step towards stabilizing the Snake River fall chinook salmon ESU simply by expanding the amount of habitat available for spawning. Draining of reservoirs under dam breaching may also reduce a major source of mortality over the long term, because predation losses in reservoirs for fall chinook salmon have been observed as high as 20%.

9.1.3 Steelhead

We do not have quantitative performance criteria for Snake River steelhead; we lack a definition of threshold escapement levels, performance criteria for recovery, and, even if criteria were quantitatively defined, we do not have an analytical method for quantifying risk. However, actions that meet jeopardy criteria for Snake River spring/summer chinook salmon would probably satisfy the biological requirements necessary for the survival and recovery of Snake River steelhead. The converse is not true; actions that would not meet jeopardy criteria for spring/summer chinook salmon would not necessarily fail for steelhead.

9.1.4 Sockeye Salmon

We do not have quantitative performance criteria for Snake River sockeye salmon. Whereas the BRWG (1994) suggested threshold escapement levels and NMFS suggested recovery levels in the Proposed Recovery Plan for Snake River salmon, we do not have an analytical method for quantifying risk to this ESU (although the current level of extreme risk is intuitively obvious). **Section 8** indicated that the population of Snake River sockeye salmon is at a critically low level, biological requirements are not met in the existing system, and survival depends upon a captive broodstock program. The conclusion of the 1995 FCRPS Biological Opinion was that sockeye salmon were not jeopardized because the hydrosystem operations were not markedly reducing the likelihood of survival and recovery of sockeye salmon (i.e., undermining the captive broodstock program). Now that the captive broodstock sockeye program is beginning to have the potential to reestablish this species in the Snake River Basin, hydrosystem options do come into play.

9.1.5 All Species

In general, PATH analyses to date identify dam breaching as the option that is, on average, more likely to meet survival and recovery population thresholds. However, the merits of breaching over transportation hinge on the extent to which smolt-to-adult returns are increased by breaching. The critical assumption that generates a marked advantage to breaching over transportation is the assumption that transported fish suffer significant delayed mortality in excess of that experienced by nontransported fish. In addition, assumptions about sources of extra mortality are also crucial to the likelihood that any management action succeeds. To date, future climate scenarios have been optimistic because they do not consider persistent poor ocean conditions.

9.2 Identifying Assumptions Critical to the Conclusion of a Marked Advantage to Drawdown

The most thorough analyses to date concern spring/summer chinook salmon. For that species, the difference in projected performance between drawdown and non-drawdown options is derived by subtracting the average probabilities that the options meet particular survival or recovery population thresholds (**Table 5.5.1.3-1**). The difference in these average probabilities represents how much more “robust” one option is than another.

Using results from analyses by the full PATH process, the most pronounced differences between the current configuration and drawdown actions were associated with the relative probabilities of satisfying the 48-year recovery criterion (Marmorek et al. 1998a). By comparing the calculated differences between A3 and A1 under clusters of alternative assumption sets, the critical uncertainties become transparent. In particular, the differences between A3 and A1 range from 48% to 16% depending on whether the analyses are based on the CRiSP or FLUSH passage model (but recall the major difference between CRiSP or FLUSH is the estimation of differential delayed transportation mortality). The choice between the Alpha versus Delta model does not appear to matter at all, and the choice between sources of extra mortality produces differences in robustness that vary from 28% to 36% (column showing "A3 – A1" in **Table 9.2-1**).

Although it appears reasonable to assign extra mortality from initial dam construction to historical declines in salmon populations, recent low return rates of Snake River salmon may result, in part, from factors other than hydrosystem effects. The most plausible suspects are ocean regime shifts, hatchery effects, and reduced stock viability. As is discussed in the next session, some of these hypotheses and uncertainties could be addressed with future research in the short term.

10.0 Opportunities for Resolving Uncertainties

10.1 Collecting More Data Under Current Conditions

10.1.1 Differential Delayed Mortality of Transported Fish

The extent to which transported fish suffer differential delayed mortality is a crucial question because the answer influences strongly the possible advantages to be accrued by dam drawdown. Ongoing direct experiments that contrast the return rates of tagged fish that pass through the hydrosystem versus the return rates of transported fish can resolve this question in a clear and unambiguous manner. It will require several years to obtain sufficient data because sample sizes of recaptured returning fish are typically so low and because delayed transportation mortality may vary with climate and measurements from only a few years may fail to capture extreme values that could have important ecological effects. Nonetheless, uncertainty about differential mortality due to transportation is a solvable problem. The importance of this uncertainty is evident in the contrast between dam drawdown options and no drawdown (A1 versus A3) assuming different D-values (**Figure 5.5.1.4-2**). Because the value of drawdown for spring/summer chinook salmon hinges on what D-values are used, one could argue that delaying action for 4 to 7 years to solidify our estimates of D may provide a better foundation for a decision. There are, however, costs to delaying. The decision must consider these costs (**Section 10.5**).

10.1.2 Effects of Ocean Conditions

Climate and oceanographic conditions clearly influence salmon survival in the oceans, and hence smolt-to-adult return rates. The PATH process to date has handled uncertainty about ocean conditions by a sensitivity analysis in which different scenarios for future climate are simulated. In the short-term, this is a sound scientific approach, and simply by broadening these sensitivity analyses we can strengthen inferences about future risks. For example PATH is now considering climate simulations in which the current (1990 through the present) condition is assumed to persist for some time into the future.

Ocean conditions are such a "black box" for all salmonids that long-term research should focus on ocean conditions and salmonid population dynamics. This research will not help inform decisions over the next few years, but could over the long term help us place population fluctuations in a broader context, so that management reactions better respond to those threats that are best mitigated by non-ocean actions. There is, however, a more fundamental scientific challenge posed by the effects of ocean conditions. It is very difficult to assign mortality (and salmonid declines) to factors such as hydrosystem effects without making some assumptions about ocean conditions. Although data regarding the marine mortality of Columbia River Basin salmonid stocks are scarce, data from other sources at least make clear how important the problem can be. Welsh (1998) calculated the average marine survival of Oregon coastal coho for three ocean regime periods: 1960 to 1977 (6.1%), 1978 to 1990 (3.3%), and 1991 to 1995 (0.5%). In 1991 and later years, average survival declined to an less than 1/5th the rate evident during the 1977 to 1990 period, and only 1/10th that during the period prior to 1977. The magnitude of these changes is more striking when one considers that for these coho stocks, there are no potential effects of "extra" or "delayed" mortality from the hydrosystem. One mechanism that may help to explain changes in SARs is the depletion of nitrate concentrations off of Vancouver Island (Welch 1998). With effects this striking (albeit for stocks outside the Snake River Basin), there is a risk of not being able to discriminate non-ocean factors against a backdrop of large variations in ocean conditions.

10.1.3 Effects of Habitat

Habitat conditions are generally good in many of the remaining areas currently supporting spring/summer chinook production within the Snake River Basin (e.g., Beamesderfer et al. 1997). PATH analyses of habitat changes have been limited to assessments of potential survival improvements that might be gained by improving habitat in these specific spring/summer chinook index areas. Whereas this is an important finding, NMFS is concerned that the full potential for increasing production of anadromous salmonids in the Snake River Basin through the restoration of habitat has not been addressed. NMFS believes that the production potential for spring/summer chinook and steelhead could be significantly expanded in the Snake River Basin through improvements in habitat coupled with conditions supporting survival in other life-history phases.

In order to promote salmon recovery through habitat restoration, a better understanding of the relationship between habitat quality and salmon and steelhead population dynamics is required. Coupling this knowledge with a thorough understanding of current habitat conditions and the likely changes to habitat quality as a result of natural disturbances and land-use related impacts will enable an accurate assessment of the role freshwater habitat can play in recovery. Key research questions include:

1. What is the relationship between habitat quality and abundance, survival and productivity of salmon and steelhead populations in the Snake River watershed?

Although researchers have previously asked this question, population levels of key species have been very low, possibly masking the influence of habitat quality on survival and productivity. Continuing to collect data on the interaction between habitat condition and fish production as population levels increase will provide a clearer indication of the role habitat plays in determining stock productivity.

2. What are likely possible future configurations of habitat and habitat quality?

Habitat quality is dynamic and continually changing. Understanding habitat quality requires an understanding of the current spatial distribution of habitat, how habitats respond to and recover from natural disturbance and how current and future land uses in watersheds are likely to alter the distribution of habitat types. Both the characteristics of the habitat and its spatial organization are key in assessing the response of fish populations.

3. What are the effects of carcass-derived organic matter and nutrients on trophic productivity of rearing habitat?

Delivery of carcass organic matter and nutrients to the Snake River watershed is about 0.2% of historical levels. The extent to which the elimination of this annual subsidy has contributed to the decline in salmon and steelhead populations is not known. Likewise, the extent to which these low levels may retard recovery is unknown. However, in other systems, materials provided by spawning salmon do substantially increase primary and secondary production, including fishes. Understanding the significance of this material in the Snake River system may assist in developing approaches to habitat and harvest management that will contribute to recovery of these depressed stocks.

10.1.4 Hatcheries and their Impact

In the PATH analyses to date, the evaluation of hatchery effects have been limited in scope. In the future, analyses can better take advantage of the fact that whereas some of the spawning and rearing index areas were exposed to significant numbers of hatchery fish, others were relatively free of hatchery influence. Increased hatchery production in the Snake River Basin may also affect the current and future productivity of wild stocks in ways that are just beginning to be manifested in productivity and returns.

10.1.4.1 Status Quo Hatchery Production

Possible adverse effects of hatchery fish on natural populations include both ecological (competition, predation, disease transfer) and genetic (loss of fitness, loss of diversity within and between populations) factors. The current lack of explicit treatment of these topics contributes to uncertainty in the overall analysis. Opportunities for ecological and genetic interactions between hatchery and wild fish depend not only on the number of hatchery fish released, but also the release locations, life-history stage released, stock histories and fish culture practices. Data of this type are available and may prove useful. For instance, spatial and temporal variation in hatchery releases might explain variation in life-cycle factors (survival and production). It is also important to realize that even if hatchery production levels remain unchanged, it does not follow that hatchery effects on productivity will also remain unchanged. If the reduction in productivity is due to genetic interactions that reduce fitness of natural populations, then, because these effects are cumulative over time, the status quo level of production would be likely to lead to a gradual deterioration of fitness with a time lag. Conversely, if the hatchery releases were terminated, one might reasonably hypothesize a gradual (although probably fairly slow) restoration of fitness over time, but also with a time lag. Matters are made more complex by realizing that there are also hypotheses for benefits due to hatchery releases – most involving reduced harvest or predation on wild fish because hatchery fish are taken instead.

10.1.4.2 Changes in the Magnitude and Nature of Hatchery Production

Snake River hatchery production (for spring/summer chinook and steelhead) increased greatly from the early 1970s through the 1980s. The increase was generally coincident with an increase in "extra mortality" for Snake River spring/summer chinook salmon. A potential negative effect of hatchery fish on wild spring/summer chinook salmon survival has been hypothesized as one of the alternative explanations for the extra mortality derived from life-cycle modeling analyses. Given the common pattern in extra mortality across the Snake River index stocks regardless of tributary-specific hatchery influence, attention has focused on interactions in the migration corridor, the barge transportation system and the lower river and estuary. Differential effects of hatcheries for Snake River stocks compared to lower river stocks may exist if:

- Negative interactions occur in the migration corridor prior to the arrival at the first dam,
- Negative interactions occur during the collection and barging of smolt from the Snake River dams, and
- The fitness of Snake River stocks is lower than those of lower river stocks.

The last factor, combined with the more demanding migration conditions of Snake River stocks, could affect fish condition and energy reserves and potentially exacerbate effects of hatchery interactions in the migration corridor or during early-ocean adaptation (Williams et al. 1998a).

Relying on statistical analyses of historical data to determine the relative likelihood of the hatchery explanation for extra mortality compared to the regime shift or hydrosystem hypotheses is difficult (SRP 1998). Progress might be made by exploring experimental manipulations of hatchery production to produce contrasts across brood years with respect to "hatchery experiences". The practical question is whether such manipulations are likely to produce unambiguous results in a timely manner. In addition to the immediate need to clarify the role of hatchery fish in contributing to the extra mortality identified through life-cycle modeling, there is also a need to evaluate more general impacts of increased hatchery production in the Snake River Basin.

Changes in hatchery programs could affect survival and recovery of listed Snake River salmonids in the following ways:

Reducing Numbers of Hatchery Fish

There is some basis for evaluating how changes in the level of hatchery production might affect natural populations. For example, Chilcote (1997) studied spawner recruit relationships in over 25 steelhead populations from Oregon and found a strong negative correlation between the percentage of naturally spawning hatchery fish and population productivity. Less crowding in the hydrosystem could affect indirect mortality through:

- Fewer ecological interactions in tributaries and ocean and
- Fewer opportunities for adverse genetic interactions with wild fish.

Changes in Type of Hatchery Programs

- Changes in rearing conditions to produce high quality smolts (i.e., healthy fish that migrate quickly to the ocean) could reduce the magnitude of hatchery-wild ecological interactions and

- Changes in broodstock practices, release strategies, etc., could affect the nature and magnitude of genetic interactions with wild fish apart from changes in numbers of fish released.

Research Needed to Address Uncertainties

Considerable scientific uncertainty surrounds most aspects of genetic and ecological interactions between hatchery and wild fish. Important research that could help resolve some of these uncertainties includes the following:

1. Determine the ecological interactions and possible effects of hatchery fish releases on wild fish. Concurrent investigation of direct social interactions between hatchery and wild juvenile salmon through a combination of laboratory and instream (or lake) studies will clarify the potential for harmful interactions. Research should examine possible detrimental effects (e.g., displacement of wild fish by hatchery fish, the transmission of disease from hatchery to wild fish, size-selective predation, the attraction of predators by concentration of hatchery fish, aggression) and suggest methods to minimize them.

2. Producing a hatchery fish with more wild-like characteristics may aid recovery of wild fish. However, a great deal of research is needed to make hatchery fish more like wild fish in morphology, body coloration, physiology, and behavior. It is critical to develop a hatchery fish that is prepared for the receiving environment and that will have increased survival to adulthood. Studies should focus on improving the operational efficiency of hatcheries both in terms of their cost efficiency and adult survival. In general, these studies should aim to improve the biological efficiency through better husbandry.

3. In many cases, conservation hatcheries will release adults and offspring from captive broodstocks. However, the reproductive success of these animals and their potential interactions with wild animals are largely unknown. Because captively reared and wild salmon experience dramatically different developmental forces, they are likely to differ in their physiology, morphology, and behavior, all of which can substantially influence their reproductive success. Comparative research on the adult reproductive behavior of captive-reared and wild salmon will elucidate potential deficiencies of captive-reared salmon and their offspring and suggest ways to mitigate for such deficiencies through improved rearing technology.

- Hatchery fish may improperly imprint during rearing or after release, potentially resulting in straying by returning adults and thus, genetic introgression on wild stocks. Research should directly address a number of concerns on the potential effects of homing and the imprinting of hatchery fish on natural gene pools and aim at providing data and hatchery management schemes to ensure that the genetic integrity of spawning stocks is maintained. s.

10.1.5 Reduced Stock Viability

As salmon populations continue to decline due to the combined effects of extrinsic factors (such as hydropower systems, reduction in freshwater and ocean habitat quality and quantity, and harvest pressure), processes intrinsic to the populations can begin to emerge as significant sources of risk. Small populations face threats to their viability due to genetic and demographic effects. For example, extinction probability in small populations is usually higher than in large populations due to chance demographic events associated with finding mates, fertility and sex ratio (Lande 1993). For populations with negative average growth rates, risks from demographic and environmental stochasticity and catastrophes are all important (Lande 1993). The viability of small populations may generally be reduced through density-dependent depensation (Lande 1998).

In addition to demographic effects, small populations face risks due to their propensity to lose adaptive, genetically-based variation and retain maladaptive variation at higher rates than larger populations (Schultz and Lynch 1997). In addition to the likely reduction in population fitness due to outright loss of genetic variation, smaller populations also can suffer reduced viability due to a increased frequency of homozygotes through inbreeding. The reduction in fitness due to inbreeding (inbreeding depression) has been demonstrated for a number of plant and animal species, including rainbow trout and Atlantic salmon (Tave 1993, Ford et al. 1999). The ultimate effects of inbreeding on population viability depend on the rate at which inbreeding occurs, the numbers of generations over which it has occurred, and the genetic history of the population. These risks need to be directly examined as potential exacerbators of hydrosystem effects.

Indicators of population size and rates of decline are straightforward conceptually, but in practice, it can be difficult to obtain good estimates of population sizes for salmonids. Expanding counts from index areas to basin-wide population estimates and factoring out the influence of hatchery-derived fish from dam, weir, or spawning survey counts can be especially challenging. Estimates of the potential genetic risks facing a population can be obtained from assessing a populations' genetically effective size from genetic or demographic information (Waples et al. 1990). In addition, genetic markers can provide useful indicators of the proportion of homozygotes in a population, which, in conjunction with studies on the fitness consequences of inbreeding, can be used to estimate population viability in small populations where inbreeding is likely.

10.2 Opportunities for Experimental Management

The most striking opportunity for experimental management comes from manipulation of hatchery programs. If hatchery production is contributing to the extra mortality associated with salmon declines, then this hypothesis can be tested by manipulating hatchery production. The problem with this particular form of adaptive management is that there are likely to be substantial timelags before it is possible to detect any effect, and even after effects are felt it is not clear what sampling effort would be required to detect improvements. Alternative experimental management strategies should be evaluated in terms of technical limits within the hatchery programs and the statistical limitations given expected precision levels.

A second more general and theoretical avenue worth exploring is the examination of possible synergistic interactions between stresses. The PATH process, and indeed almost all risk analyses, examine risks in a linear additive framework. Recent thinking in ecology suggests that ecosystems are most threatened when exposed to compounded perturbations (Paine 1998) and that such additive modeling can underestimate the threats to populations. For salmon this means that experimental management aimed at creating different combinations of risks would be useful for exploring possible compounding of threats. A specific example is the accelerating decline of salmon that may be induced when stocks decline below a low level due to hydrosystem or ocean conditions, only to be further driven down by genetic degradation of stock viability and reduced habitat quality as a result of the absence of nutrient inputs. Experimentally, it might be feasible to artificially add salmon carcasses as a way of elevating nutrient inputs.

Dam drawdown is itself experimental management in the sense it is reversible (or at least not as irreversible as is extinction). It is important that if dam drawdown is pursued, careful population monitoring be conducted that is aimed more generally than at assessing whether survival and recovery criteria outlined by PATH are met. If drawdown is implemented, detailed data must be collected to see whether salmon are on an improving trajectory (even if the improvement may not be as dramatic or as rapid as is desired). In general, experimental adaptive management requires monitoring programs that can detect the presence or absence of a system's response. For salmonid monitoring, this means greater attention will need to be given to the problem of sampling error than has been traditional in run reconstruction analyses.

10.3 Research Milestones for Reducing Uncertainty

This report should make it clear that there are important uncertainties that have substantial consequences for decisions about alternative management actions. It is also clear that research can help resolve some of these uncertainties. But research involves delay, and delay involves risk. For this reason, it is useful to identify particular research goals and the likely time it would take to meet those goals. Below are critical milestones for research efforts with an estimated time horizon for meeting each of those challenges.

<u>Reducing Uncertainty regarding D-values</u>: Because a PIT-tag experiment was initiated 5 years ago, data are already being accumulated to strengthen the confidence in our estimates of D-values. Now it is simply a matter of spanning enough years to obtain larger sample sizes and to sample a range of environmental conditions. This research should meet its challenge within 5 to 10 years.

<u>Exploring Other Factors Contributing to Fish Passage Mortality</u>: Differential delayed transportation mortality is NOT the only issue surrounding mortality and passage through the hydrosystem. There are several other subtleties and possibilities that warrant study (if the dams are NOT breached), which include the impact of timing, the possibility that fish of different intrinsic fitness are selected by different transportation routes, and details of the effects of passage route on mortality. These issues can be answering using the basic PIT-tag technology recently implemented by NMFS and could also yield significant answers within 5 to 10 years.

<u>Developing A Cohesive Framework for Measuring the Risk of all "four H's's" in the same Currency</u>: In assembling this report it became clear that although hatcheries, habitat and harvest have been examined through the PATH process, they have not been examined with the same clear currency (i.e., linking specific changes in fish population dynamics to management actions) as have hydrosystem effects. There are less data for these other "H's" than there are for the hydrosystem and hence, a simpler model will be more appropriate. NMFS is in the process of developing an analytical framework that places all of the H's on equal footing, so that their

contributions to recovery can better be managed and balanced with one another. The time scale for this research initiative is on the order of 1 to 2 years.

The Link Between Habitat Improvements and Fish Recovery: Now that GIS data bases with habitat attributes are becoming available, it is becoming increasingly feasible to start a more quantitative assessment of options for salmon recovery through habitat improvements. The statistical relationships between habitat factors and fish populations can be developed rapidly (1 to 2 years), but there will also be a need for independent empirical work, because much of the habitat description in these data bases is still quite crude. The empirical research for habitat restoration will involve direct evaluation of ongoing restoration projects and will require 10 to 20 years to yield definitive data (although obviously some results could become apparent in much less time).

The Link Between Hatchery Practices and Fish Populations: In order to evaluate interactions between hatchery practices and wild fish, a variety of models, direct small-scale experiments, and adaptive management options, are being considered. This is more difficult than simply tabulating the number of hatchery releases because alterations in hatchery production techniques can alter the type of fish being released. Statistical and modeling advances can be made on the time scale of 1 to 2 years, but the important empirical work will require 10+ years.

The role of "other factors" in Extra Mortality: Many factors other than the four H's effect salmonids. The most notable is probably the change in predation pressure associated with increasing marine mammal populations, concentrations of Caspian terns and other birds in the Columbia River, and predation by different fish species. Understanding both the importance of this predation and options for minimizing its impact will require 5 to 10 years, with many of the studies already underway.

General Impacts of Ocean Conditions and Regime Shifts: The most challenging research questions surround the impact of ocean conditions. Partly the challenge arises because researchers must wait for the ocean conditions to follow their own cycles of variation in order to assess consequences. A second challenge arises because tracking the fate of fish from different stocks, once they are in the ocean, is so difficult – even with PIT-tag technology. Opportunities are emerging for remote sensing and other technological improvements to stimulate progress in this research area, but uncertainty about ocean conditions and salmon population recovery will require decades of research to resolve (20+ years).

10.4 Potential Surprises Outside the Risk Framework

Life-cycle modeling analyses should incorporate consideration of increased mortality during the implementation of drawdown in the lower Snake River. We discuss these possible risks, below.

10.4.1 Direct Risks Due to Construction Activities

10.4.1.1 Juvenile Passage

Any construction activity at mainstem dams has the potential to affect juvenile salmonids during their outmigration. Yearling spring/summer chinook salmon and steelhead migrations typically end prior to the August-September construction window. However, ocean-type subyearling fall chinook salmon migrate through December. Thus, NMFS' primary concern for juvenile salmonids during construction is the passage conditions fall chinook salmon will encounter as they pass through turbines operated at heads below the turbine design and optimum efficiency ranges. NMFS concludes there would be a potential for substantial mortality to fall chinook salmon outmigrants passing through turbines during construction.

10.4.3.2 Adult Passage

NMFS participated in the development of temporary fish passage concepts during construction. The recommended option calls for trapping adults at Ice Harbor Dam and transporting them in trucks to above Lower Granite Dam. While this is probably the best option, trap and haul operations entail some risk to the stocks. These include potential losses associated with transporting adults when water temperatures during August can reach 70° F or higher, handling losses, disease transmission, and the inappropriate transportation of Columbia River strays to above Lower Granite. These types of systems typically need an initial period of operation where problems are identified and corrected. Little time will exist for fine tuning the new trapping facilities.

Short-term blockages to adult passage may occur after construction has been completed. The breach at each dam has been sized to maintain adult passage water velocity criteria of 1.5 to 4.0 fps for passage during low to moderate river discharges. During high flow events where river flows exceed 170,000 cfs, average velocity will be greater than 8 fps. Because the breach is an extended run, these high passage velocities could cause delays to listed spring/summer chinook salmon, sockeye salmon, and steelhead. The Corps has investigated the placement of large boulders to facilitate adult passage under high flow conditions. However, the placement, stability, and effectiveness of these structures have not been hydraulically modeled, nor have the proposed breach configurations.

In addition, blockage of adult entry into tributaries may occur where sediment deltas have formed over the years of impoundment. The smaller tributaries in the reach from Ice Harbor Dam to Asotin would be affected. Adult entry into these small streams could be blocked by the shallow, braided flow until a new channel is established by erosion.

Finally, during the first year(s) of operation after breaching, high turbidity levels may impede the upstream migrations of adult salmon. Radio tagged adults have been observed to stop or slow their migration during high flow, high turbidity events. The ambient turbidity levels presented to adult fish and their behavioral response would be highly variable and may create problems.

10.4.2 Sedimentation Risks Due to Construction Activities

Turbidity

Drawdown is likely to increase sedimentation, which could have several potential impacts on Snake and Columbia River salmonids. Sedimentation impacts would vary based on the type and timing of dam removal and salmonid life stage. Although several studies indicate that increased sedimentation/turbidity is likely to have deleterious effects on Pacific salmon at each of their life-history stages, the impacts from increased sedimentation from drawdown would be relatively short term (Alonso et al. 1988, Beschta and Jackson 1979, Bjornn and Reiser 1991, Carling 1984, Carling and McCahon 1985, Chapman and McLeod 1987, Chapman 1988, Diplas and Parker 1985, Einstein 1968, Lisle 1980, Marcus et al. 1990, Olsson and Persson 1988, Reiser and White 1988, Sigler et al. 1984, Young 1989, Young et al. 1990, Young et al. 1991).

10.4.2.2 Chemical Contaminants

There are concerns regarding contaminants associated with impounded sediments that have accumulated behind dams in the lower Snake River complex. The sediment released could be potentially contaminated with chemicals that can have deleterious effects on salmonids and their prey. The more recently deposited sediments in the impound area will be the first to flow downstream and be deposited, meaning the older, and potentially more contaminated sediments that are deeper down in the impound area, will flow downstream latter and be deposited on top of those newer sediments. The deeper sediments in the impound areas could be more highly contaminated, because environmental controls (e.g., the Clean Water Act) on pollutant discharges were less stringent in the previous decades when deposition had occurred. Therefore, any potential effects from sediment-associated chemical contaminants could increase with time. Because the chemical contaminant data are currently not available, a full assessment of potential impacts on salmonids and their prey is not possible. The ability to assess the risk of toxic chemicals on growth and survival of salmonids in-river will be dependent on a number of factors, however, the quality of the analytical data for the concentrations of contaminants in sediments from behind the dams will be an important factor.

10.5 Cumulative Risk Assessment

The PATH analyses and NMFS assessment to date have taken up each threat to salmonids, one threat-at-a-time: harvesting, hatcheries, transportation mortality, upstream blockage, ocean conditions, water quality, habitat quality and so on. The results are integrated using a life cycle model that tracks fish though the hydrosystem and their entire life cycle in an explicit quantitative fashion. One limitation of this approach is that “threats” or “stresses” are not necessarily additive or linear. Analogies from human risk analyses are illustrative - the risk of smoking is greatly exaggerated by also engaging in heavy alcohol consumption. Similar principles are likely to apply to the health of populations and ecosystems. Cumulative risk analysis, which deals with such synergism and nonlinearity in risks, is a relatively new scientific field (NRC 1996a). It should be applied to salmonids in the Snake River if we hope to optimize our recovery efforts. Recent research indicates that ecological disturbances and threats exert substantially greater total effects than one would estimate from simply summing together the independent risks (Paine 1998). But to conduct cumulative risk analysis, the mechanism underlying interactions between threats have to be identified and understood. For salmonids, this will require a broader view of risk analysis than has been traditional, and will require that research regarding habitat, hatcheries, harvest, and hydrosystem are better integrated into one framework (which is also a recommendation of NRC [1996b]).

NMFS has initiated a cumulative risk framework that is developing simpler models than have been used by PATH to date, but entail the same core life-cycle model. The simplicity is obtained by eliminating the detailed flow/passage models (which are appropriate when focusing on hydrosystem actions, but makes less sense in the context of a broader risk assessment). The baseline demography for different salmonid stocks will be estimated using the core life-cycle model and then alterations in this demography will be connected to variations in hatchery, harvest, habitat, and hydrosystem factors. The connection can be either statistical, where the data exist, or speculative in the form of plausible hypotheses to be tested. By placing the effects of the "four H's" in one simple and common life-cycle model, it will be much more straightforward to evaluate how suites of policy actions (suites of variations in the four H's) are likely to influence salmonid populations.

10.6 A Bottom Line: Balancing Uncertainties with Actions

Any scientific decision analysis has uncertainties. Scientists cannot make the decisions, they can only illuminate likely outcomes of actions and the uncertainties associated with those predicted outcomes. In this report the major actions compared are breaching of the lower Snake River dams versus enhanced transportation and no breaching. If all assumptions are equally weighted and PATH estimates of delayed transportation mortality are used, breaching is clearly much more likely than current operations to meet survival and recovery population thresholds. However, the story is not as simple as the preceding sentences suggest. Importantly, gaps in the data and major scientific uncertainties complicate the clarity of the preceding remarks.

Critically, using the equally-weighted assumptions passed through the PATH process, breaching increases the average probability of meeting recovery population thresholds by 30% for spring/summer chinook salmon. However, if the PATH prospective models are run assuming only higher D-values (minimal differential delayed mortality due to transportation), this difference and the advantages to breaching are substantially reduced and may even disappear under certain assumptions about extra mortality. The key issue then concerns the cost of delaying to resolve the uncertainty about transportation mortality (or D-values) and the primary causes of extra mortality. Research is already underway for directly estimating transportation mortality, and adequate data should be forthcoming within less than 10 years. However, information regarding sources of extra mortality will require research that may begin to yield useful information in as few as 5 years, but ultimately is likely to require 10 to 20 years before major reductions in uncertainty are realized. During that time period, it is the "survival" of stocks that are most likely to be put at risk (**Table 5.5.3.1-1**). Based on existing PATH analyses, the cost of this delay <u>is at least</u> an additional 8% risk in failing to stay above population survival thresholds over a 24-year interval for spring/summer chinook salmon.

In general, alternative management actions for the hydrosystem need to be examined in light of a broader risk analysis that recognizes a combination of factors which may compound one another in placing salmon at risk. Moreover, there is a need to recognize that science does not point to any magical easy solutions. Indeed, there are plausible, pessimistic scenarios (such as persistent poor ocean conditions) that make forecasts of salmon survival and recovery under any single management action unlikely. The "bottom line" then is a conclusion with caveats: although science cannot say that breaching will certainly yield salmon recovery, science does make it clear that breaching is the most risk-averse management option. The value of that risk averseness as quantified in this report is a question for policy-makers, and not science.

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Table 3.4-1: Hydrosystem management actions examined by PATH. The A6 and A6' options have not yet been quantitatively defined. An "X" indicates the management action is implemented; a "--" indicates no action.

Scenario	Flow Augmentation		Drawdown of Four Snake River Dams	Drawdown 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[4]
A3	X	X	Natural River	--	--	--
A6	X	X[5]	--	--	--	X
A6'	--	--[6]	--	--	--	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) A2' maximizes transportation using current system configuration plus system improvements such as surface bypass collectors which would promote transportation of a larger proportion of the run.

(5) A6 includes the flow augmentation programs specified in the 1995 and 1998 FCRPS Biological Opinions for the Columbia and Snake rivers plus an additional 1 million acre-feet from the upper Snake River Basin.

(6) A6' includes continuation of the flow augmentation programs in the 1995 and 1998 FCRPS Biological Opinions except for the 427,000 acre-feet delivered from the upper Snake River Basin. Flow augmentation water would continue to be supplied from storage reservoirs in the upper Columbia and from Dworshak Reservoir in the Clearwater Subasin.

Table 4.3.2-1: Spill cap volumes, limiting factors, and hours of spill at lower Snake River dams under the 1998 Supplemental FCRPS Biological Opinion (NMFS 1998).

Project	Spill Volume (kcfs)	Limiting Factor	Hours of Spill
Lower Granite	45	Gas Cap	6 p.m.-6 a.m.
Little Goose	60	Gas Cap	6 p.m.-6 a.m.
Lower Monumental	40	Gas Cap	6 p.m.-6 a.m.
Ice Harbor	Night: 75 Day: 45	Night: - Gas Cap Day: - Adult Fish Passage	24 hours

Table 4.4.1-1: Glossary of frequently used technical terms.

Term	Definition
Assumption sets	When running the life-cycle model to generate future salmon population levels, several choices must be made regarding the magnitude of particular sources of mortality, routes of fish passage, flow rates, and so on. A complete set of these assumptions, used to generate 4,000 replicate Monte Carlo simulations of the effect of an alternative hydrosystem management action, is called an assumption set.
BKD	A disease of salmonids caused by the bacterium *Renibacterium salmoninarum*. The bacterium can be passed between juvenile fish where they are concentrated in hatcheries and in transportation systems and can be passed to the next generation by an infected female.
Conversion rates	The estimated survival of adults during upstream migration is expressed as a "conversion rate". Conversion rates are calculated by dividing the count of a particular group of adult fish at the uppermost dam by the count of that group at the lowest dam, subtracting out estimates of harvest and tributary harvest between the dams (see formula in **Section 5.2.2**).
CRiSP	Acronym for **C**olumbia **Ri**ver **S**almon **P**assage, the passage model developed by the Center for Quantitative Studies at the University of Washington under contract to the Bonneville Power Administration.
Differential delayed transportation mortality	Additional mortality suffered by transported fish after their release from the transport vehicle into the Columbia River below Bonneville Dam – hypothesized to be caused by stresses associated with the transportation system. Differential mortality is measured as the ratio of the post-Bonneville-Dam survival of transported fish to that of nontransported fish. "Delayed" transportation mortality is differentiated from any "direct" mortality of fish that occurs during transportation.
D-values	Measure used to quantify differential delayed transportation mortality. A D-value of 1.0 would mean that there was no differential delayed transportation mortality (there could be mortality, it is just not different between transported and non-transported fish). The lower the value of D (relative to 1.0), the larger the differential delayed transportation mortality. It is possible for D to be greater than 1 (in which case transported fish would have survived at a higher rate than non-transported fish).
Extra mortality	"Extra mortality" is any mortality occurring outside the migration corridor (i.e., below Bonneville Dam) that is not accounted for by in-common climate effects or by differential delayed transportation mortality.
FLUSH	Acronym for **F**ish **L**eaving **U**nder **S**everal **H**ypotheses, the passage model developed by the states of Oregon, Washington, and Idaho and the Columbia River Intertribal Fish Commission.
Ocean regime shift	Cycle of oceanographic conditions that alters patterns of circulation, the distribution of predators and prey, and productivity. Cycles have been observed on the timescale of years (El Niño), decades (Pacific interdecadal oscillations), and thousands of years (ice ages) (**Section 4.4.3.2**). The current ocean regime, and a shift on the timescale of years or decades, may affect the likelihood of recovery under any of

	hydrosystem management alternative.
Passage model	Mathematical simulation of the effect of downstream passage (through eight Federal mainstem hydro projects) on the survival of juvenile salmonids. PATH used two passage models, CRiSP and FLUSH (see above). The models differ both in their mathematical structure and in assumptions about survival through various parts of the hydrosystem (see page 25 in Marmorek and Peters (1998) *(March 1998 report)* for a brief comparison.
Recovery	The process by which the ecosystem is restored so it can support self-sustaining and self-regulating populations of listed species as persistent members of the native biotic community. This process results in improvement in the status of a species to the point at which listing is no longer appropriate under the ESA.
Risk averse	In the context of PATH analyses, "risk averse" corresponds to a management action that minimizes the risk of <u>not</u> meeting recovery and survival criteria, an action that succeeds in satisfying performance criteria over the widest range of assumptions.
Survival	The species' persistence beyond the conditions leading to its endangerment, with sufficient resilience to allow for potential recovery from endangerment. The condition in which a species continues to exist into the future while retaining the potential for recovery.

Table 5.1.1-1: Changes in the number of spawning stream-type chinook salmon, and contribution of biomass, nitrogen and phosphorus from their carcasses to the Snake River watershed. Biomass values assume average chinook salmon body weight is 10 kg. Input values for N and P assume that nitrogen constitutes 3.04% and phosphorus 0.36% of wet body weight in Pacific salmon (Larkin and Slaney 1997).

Material	Historic Levels	Early 1960s	Current
Spawners/year	1.5 million	140,000	3,000
Biomass (MT[1]/year)	15,000	1,400	30
Nitrogen (MT/year)	456	42.5	0.91
Phosphorus (MT/year)	54	5.0	0.11
[1] Metric Tons			

Table 5.5.1.1-1: Set of assumptions and alternative values for these assumptions, used in the PATH analyses.

Uncertainty	Hypothesis Label	Description
Uncertainties/hypotheses related to downstream passage to Bonneville Dam		
In-river survival assumptions – Passage Models	PMOD1	CRiSP direct survival estimates.
	PMOD2	FLUSH direct survival estimates.
Fish Guidance Efficiency (FGE)	FGE1	FGE w/ESBS > FGE w/STS. (ESBS = extended-length submersible bar screens). (STS = standard-length submersible travel screens).
	FGE2	FGE w/ESBS = FGE w/STS.
Historical/Turbine + Bypass Survival	TURB1	Turbine survival = 0.9. Bypass survival = 0.97 - 0.99, depending on the project.
	TURB4 TURB 5 TURB 6	Various mechanisms for turbine/bypass survival during some historical years. Survival is lowest under TURB4, and highest under TURB5.
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
	PRER2	8 years
Equilibrated Snake River juvenile survival rate under drawdown	EJUV1	0.85
	EJUV2	0.96
Duration of Transition Period after drawdown	TJUVa	Survivals reach equilibrated values 2 years after dam removal.
	TJUVb	Survivals reach equilibrated values 10 years after dam removal.
Other uncertainties/alternative hypotheses		
Transportation models	TRANS1 or T1 (FLUSH only)	Relationship established between TCR and FLUSH in-river survival, 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.
	TRANS2 or T2 (FLUSH only)	TCRs derived from TRANS1 adjusted by 0.83 to reflect poorer survival of transported fish from last dam to spawning grounds. *(Note: not used in analyses)*

Uncertainty	Hypothesis Label	Description
	TRANS3 or T3 (CRiSP only)	For pre-1980 retrospective analyses, relative post-BONN survival set at average D-value estimated from seven T:C studies in 1970's and associated CRiSP in-river survival rate estimates. Post-1980 retrospective analyses use average 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 1980 values.
Distribution of Extra Mortality	ALPHA	Extra mortality is specific to each sub-region, and affected by climate variables.
	DELTA	Extra mortality is independent of the common year effects which affect several subregions.
Extra mortality/Future climate	EMCLIM1	Extra mortality is here to stay; future climate is sampled from historical distribution with autoregressive properties.
	EMCLIM2	Extra mortality is here to stay; future climate follows cyclical pattern.
	EMCLIM3	Extra mortality is proportional to hydrosystem-related mortality, future climate is sampled from historical distribution with autoregressive properties.
	EMCLIM4	Extra mortality is proportional to hydrosystem-related mortality, future climate follows cyclical pattern, with both long (60-year) and shorter (18-year) cycles.
	EMCLIM5	Both extra mortality and future climate follow cyclical pattern.
Habitat Effects	HAB0	Same management as current.
	HABB	Implementation of all possible habitat restoration or protection.

Table 5.5.1.1-2: Summary of estimates of duration, juvenile survival, and adult survival for each of the four time periods.

Time Period	Duration (Years)	Juvenile Survival[1]	Adult Survival[2]
Pre-removal	3 years or 8 years	Determined by passage models	Current estimates
Removal	2 years	No change from pre-removal period	No change from pre-removal
Transition	2 years or 10 years	Linear increase from pre-removal survival to equilibrated survival	Linear increase from pre-removal to equilibrated value
Equilibrium	Determined by length of simulation period	0.85 or 0.96	0.97

[1] Juvenile survival is calculated over the four Snake River project reaches.
[2] Conversion rates.

Table 5.5.1.3-1: Average probabilities (across all, equally weighted assumption sets) of exceeding survival and recovery escapement levels for spring/summer chinook salmon for alternatives A1, A2, A2', A3, and B1. Analyses for A3 assume 3-year and 8-year delays prior to dam breaching, respectively (Marmorek et al. 1998). The number in parentheses indicates the sample size used to calculate each average.

Action	24-Year Survival	48-Year Recovery
A1	0.65 (240)	0.50 (240)
A2	0.64 (240)	0.47 (240)
A2'	0.65 (240)	0.48 (240)
A3 (3-year delay)	0.73 (439)	0.82 (439)
A3 (8-year delay)	0.69 (439)	0.82 (439)
B1	0.71 (240)	0.85 (240)

Table 6.2.1-1: **Availability of coded wire tag (CWT) data for estimating ocean exploitation rates (by stock group).**

Natural Fall Stock	Natural CWT Group	Hatchery CWT Group
SRB	--	Lyons Ferry BY 1984-1989, 1991
HYURB	Hanford wild BY 1986-1991	Priest Rapids BY 1975-1991
DES	Deschutes BY 1977-1979 distribution comparison	Lyons Ferry BY 1984-1989, 1991
NFL	North Fork Lewis wild BY 1977-1979, 1982-1991	--

Table 6.2.1-1: Subbasin exploitation rate and mainstem conversion and exploitation rates used to expand natural SRB escapement to the Snake River area spawning grounds and fisheries to recruits at the Columbia River mouth. Ocean exploitation rates were used to expand Columbia River-mouth recruits to account for impacts of ocean harvest.

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
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
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

Table 6.4.1.2-1: Cutoff dates for transporting fall chinook salmon smolts at Lower Granite (LGR), Little Goose (LGO), Lower Monumental (LMO), and McNary (MCN) dams.

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

Table 6.4.2-1: Future in-river harvest rates for the SRB and HYURB stocks based on recruits to the river mouth that were used in the PATH modeling analysis.

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

Table 6.5.1-1: Average probabilities (across all, equally weighted assumption sets) of exceeding survival and recovery escapement levels for fall chinook salmon for alternatives A2, A2', A3, and B1 (Marmorek et al. 1998). The number in parentheses indicates the sample size used to calculate each average.

Action	24-Year Survival	48-Year Recovery
A2	0.90 (96)	0.28 (96)
A2'	0.91 (96)	0.32 (96)
A3	0.95 (384)	0.99 (384)
B1	0.95 (576)	0.99 (576)

Table 7.5-1: 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)	0.010 – 0.012 (0.011)
Necessary Incremental Change (Historical Mean ÷ Recent Mean)	6.9x	3.5x

Table 7.5-2: 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)	0.012 – 0.015 (0.013)
Necessary Incremental Change (Historical Mean ÷ Recent Mean)	11.2x	4.2x

Table 8.2.2.1-1: Radio-telemetry estimates of per-project survival over the four-project reaches in the lower Columbia and lower Snake rivers for adult spring/summer chinook salmon and steelhead, respectively (C. Ross, Fishery Biologist, NMFS, pers. comm., February 23, 1999).

River Reach	Per-Project Survival (Adults)	
	S/S Chinook	**Summer Steelhead**
Lower Columbia (BON – MCN)	97.4%	98.8%
Lower Snake (IHA – LGR)	95.9%	95.5%

Table 8.4.1-3. Fish guidance efficiencies for sockeye salmon at John Day and McNary dams. Note: Tests with standard- versus extended-length screens took place years apart. A direct comparison of the results may also be confounded by factors such as high flows in 1996.

Project	Test Dates	Screen Type	Op.Gate Position	Average Percent Fish Guidance Efficiency				No. Observ.
				Sockeye Salmon	Yrlng Chinook	Steel-head	Subyrlng Chinook	
MCN	May 18-21, 1987	LSTS [1] (33 in.)	ROG [4]	Slot5A-**73%** Slot5B-**71%**	84% 84%	85% 83%	18% 21%	n = 4
JDA	May 8-23, 1985	STS [2]	NOG [5]	Slot7B-**41%**	72%	86%	21%	n = 5
JDA	May 8-25, 1996	ESBS [3]	NOG	Slot7B-**80%**	83%	95%	--	n = 16

[1] LSTS = lowered submersible traveling screen
[2] STS = standard-length traveling screen
[3] ESBS = extended-length submersible bar screen
[4] ROG = raised operating gate
[5] NOG = no operating gate

Table 8.4.1-2: Rates of descaling percent for sockeye salmon / kokanee as observed at lower Snake River and McNary dams.

Date	Stock Origin	Lower Granite	Little Goose	Lower Monumental	McNary	Notes
1997	Hatchery sock	9.9	0	13.9	9.7	
	Wild sock	24.5	10.7	14.1	18.7	(1)
1996	Hatchery sock	3.8	5.3	6.7	11.6	
	Wild sock	18.4	14.8	5.9	11.5	
1995	Hatchery sock	3.2	9.4	4.8	5.7	
	Wild sock	30.1	15.7	13.6	18.3	
1994	Hatchery sock				7.8	
	Wild sock	12.5	15.1	21	12.4	
1993	Hatchery sock			26.6	2.9	
	Wild sock	27.3	11.1		8.5	
1992	Combined	2.3	6.6		13.1	(2)
1991	Combined	0.5	5.9		10.8	(3)
1990	Combined		10			
1989	Combined				16.8	(4)
1988	Combined				10.4	
1987	Combined				10.9	
1986	Combined				21.1	
1985	Combined				8.8/3.0	(5)
1984	Combined				10.8	
1983	Combined				9.8	
1982	Combined				14.6	
1981	Combined				5.7-31.4	(6)

(1) There have been nearly no wild sockeye salmon in the Snake River system in recent years. Wild sockeye salmon at lower Snake River projects (Lower Granite, Little Goose, and Lower Monumental dams) were probably anadromous offspring of residual matings or anadromous kokanee, the latter possibly from Dworshak Reservoir.
(2) Prior to 1995, combined (hatchery + wild) observations at lower Snake River projects probably included hatchery sockeye salmon and wild anadromous kokanee, as above.
(3) 1993 – 1997 reported in annual reports of the Juvenile Fish Transportation Program. Numerous authors. U.S. Army Corps of Engineers, 1995 – 1998.
(4) 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.
(5) Descaling criteria, developed by the Fish Transportation Oversight Team, changed in 1985. Criterion = 3.0 during earlier period; raised to 8.8 after 1985.
(6) 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]).

Table 9.2-1: Percent difference in relative probabilities of exceeding 48-year escapement levels under dam breaching (A3) versus the current system (A1).

Hypothesis	**A1**	**A3**	**A3 – A1**
FLUSH	38.3	86.4	48.1
CRiSP	62.7	78.5	15.8
Alpha model	50.2	82.1	31.9
Delta model	50.8	82.7	31.9
BKD extra mortality	37.9	74.3	36.4
Hydro extra mortality	59.6	88.9	29.3
Regime-shift extra mortality	57.5	85.8	28.3

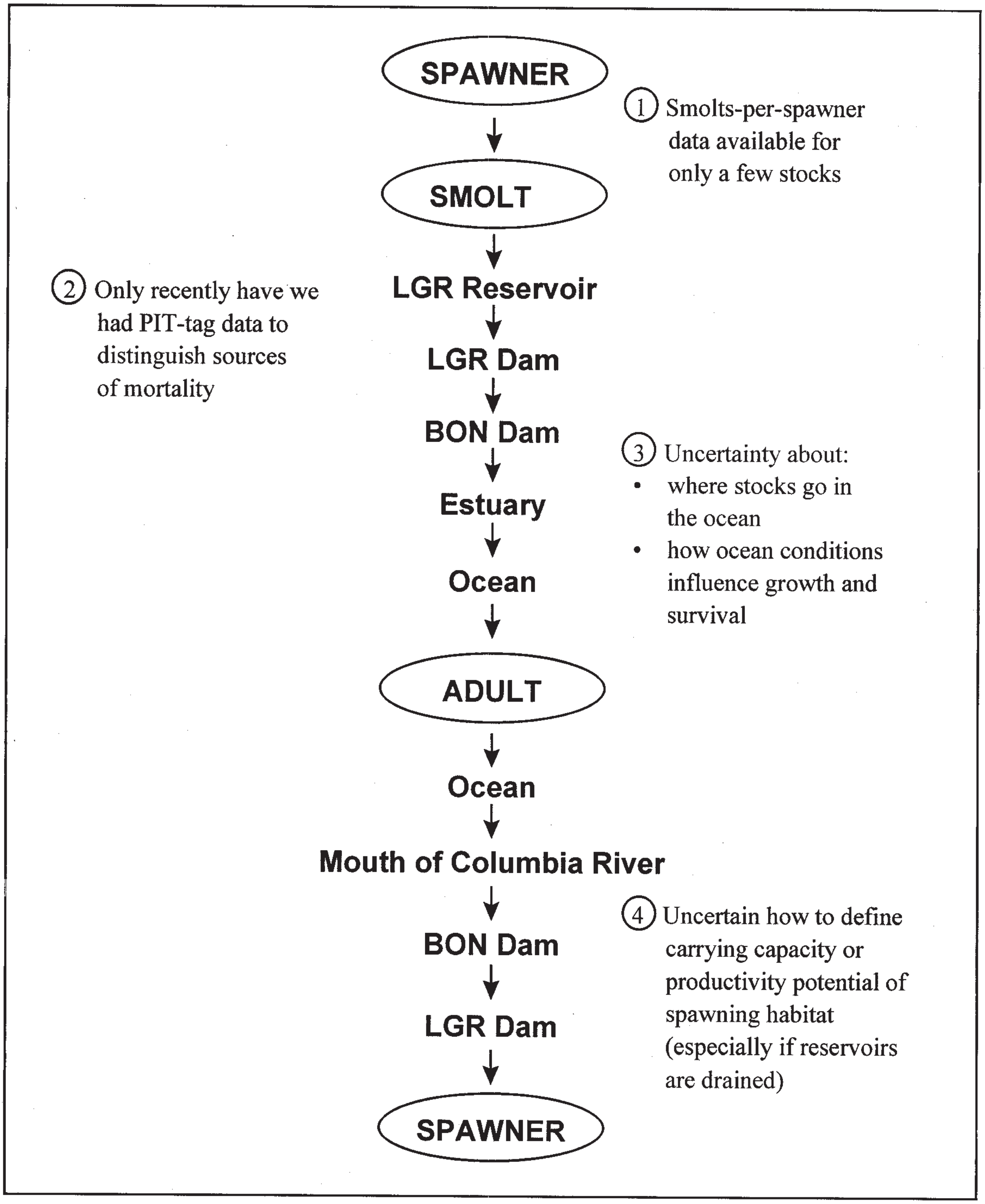

Figure 4.4.1-1: Straight-line representation of a generalized life cycle of Snake River salmonids. Notes show examples of points in the life cycle where empirical data are missing or incomplete. In the absence of complete information, both NMFS and PATH must make assumptions about quantitative changes in survival at these steps.

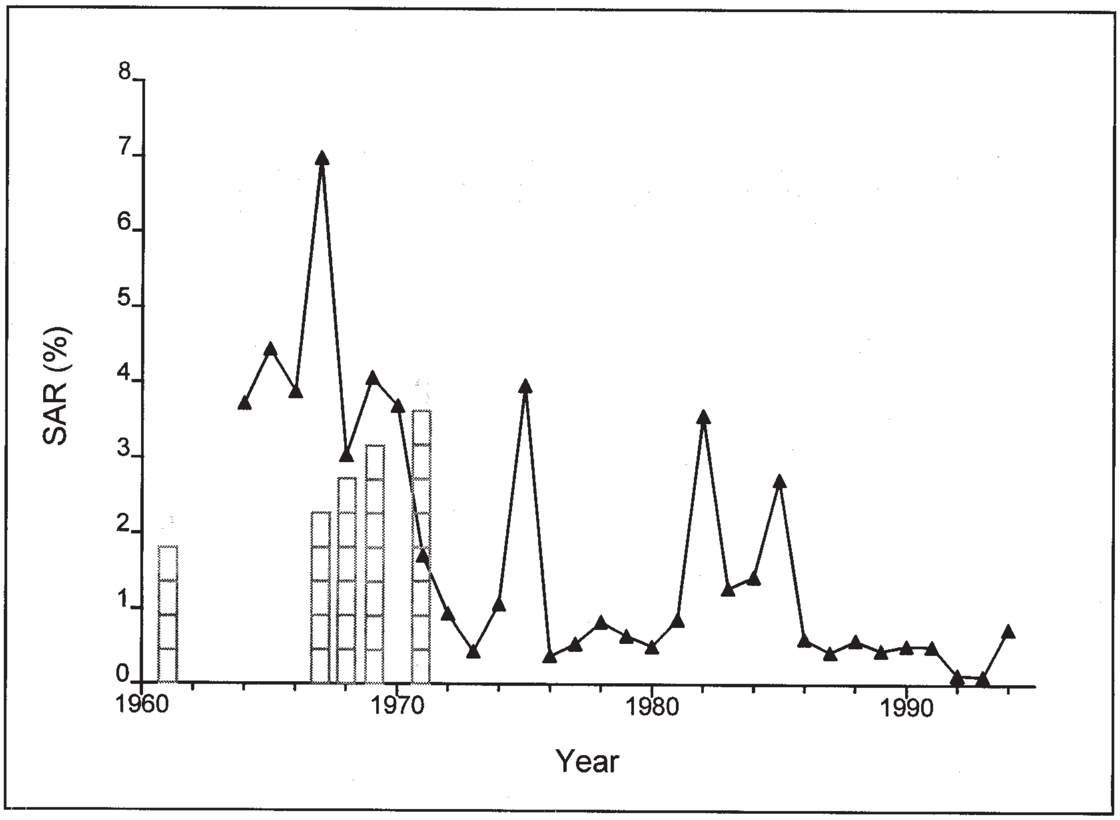

Figure 4.4.1-2: Coincidence in time of the development of the hydrosystem (cumulative number of mainstem (lower Snake and lower Columbia River) dams and the onset of low smolt-to-adult return rates (SARs) for wild spring/summer chinook salmon (Williams et al. 1998b). Smolt-to-adult return rates include escapement to the upper-most dam plus harvest.

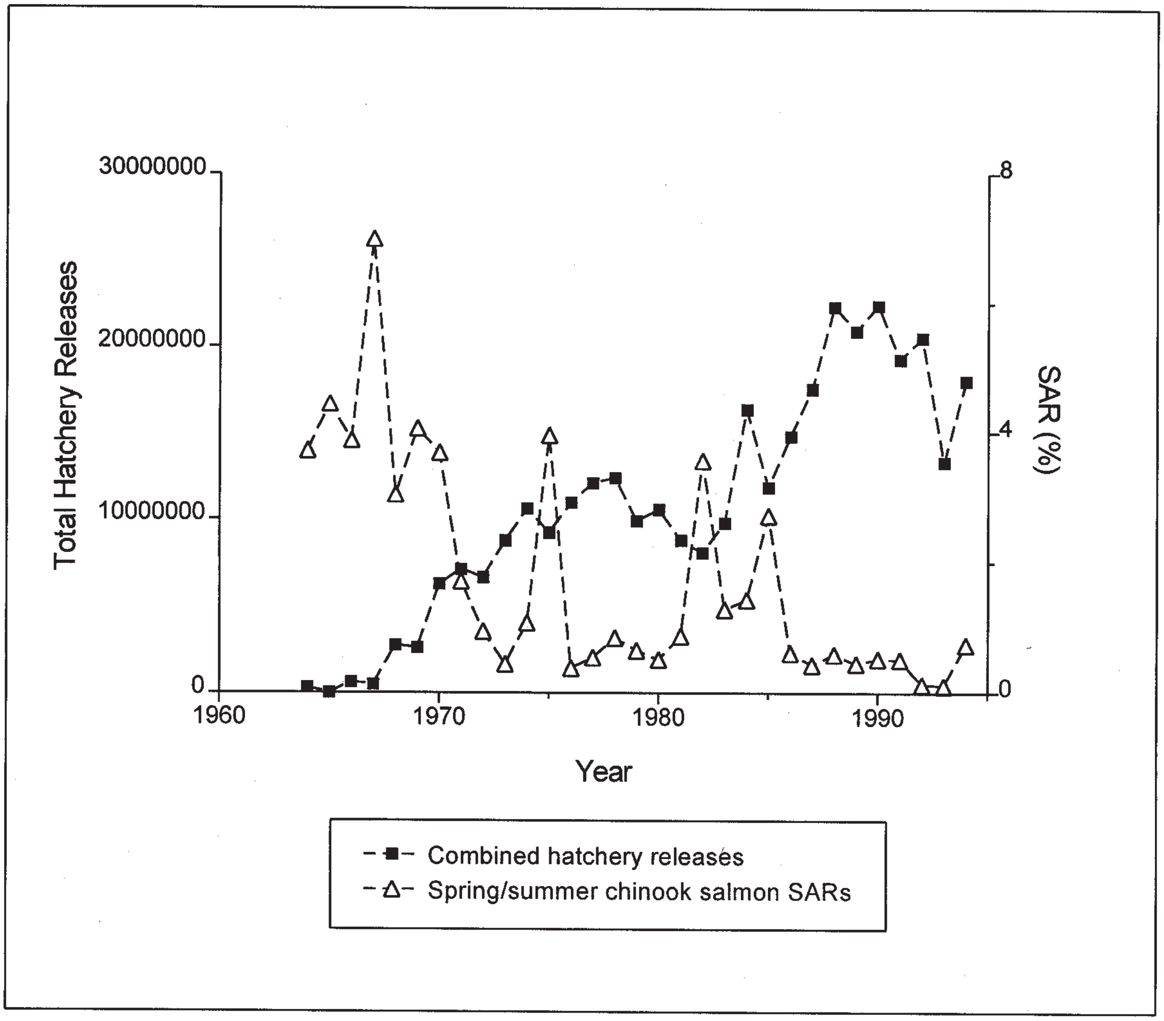

Figure 4.4.1-3: Coincidence in time of hatchery releases (combined releases of spring/summer chinook salmon and steelhead; Williams et al. 1998a) and the onset of low smolt-to-adult return rates (SAR) for wild spring/summer chinook salmon (Williams et al. 1998b). Smolt-to-adult return rates include escapement to the upper-most dam plus harvest.

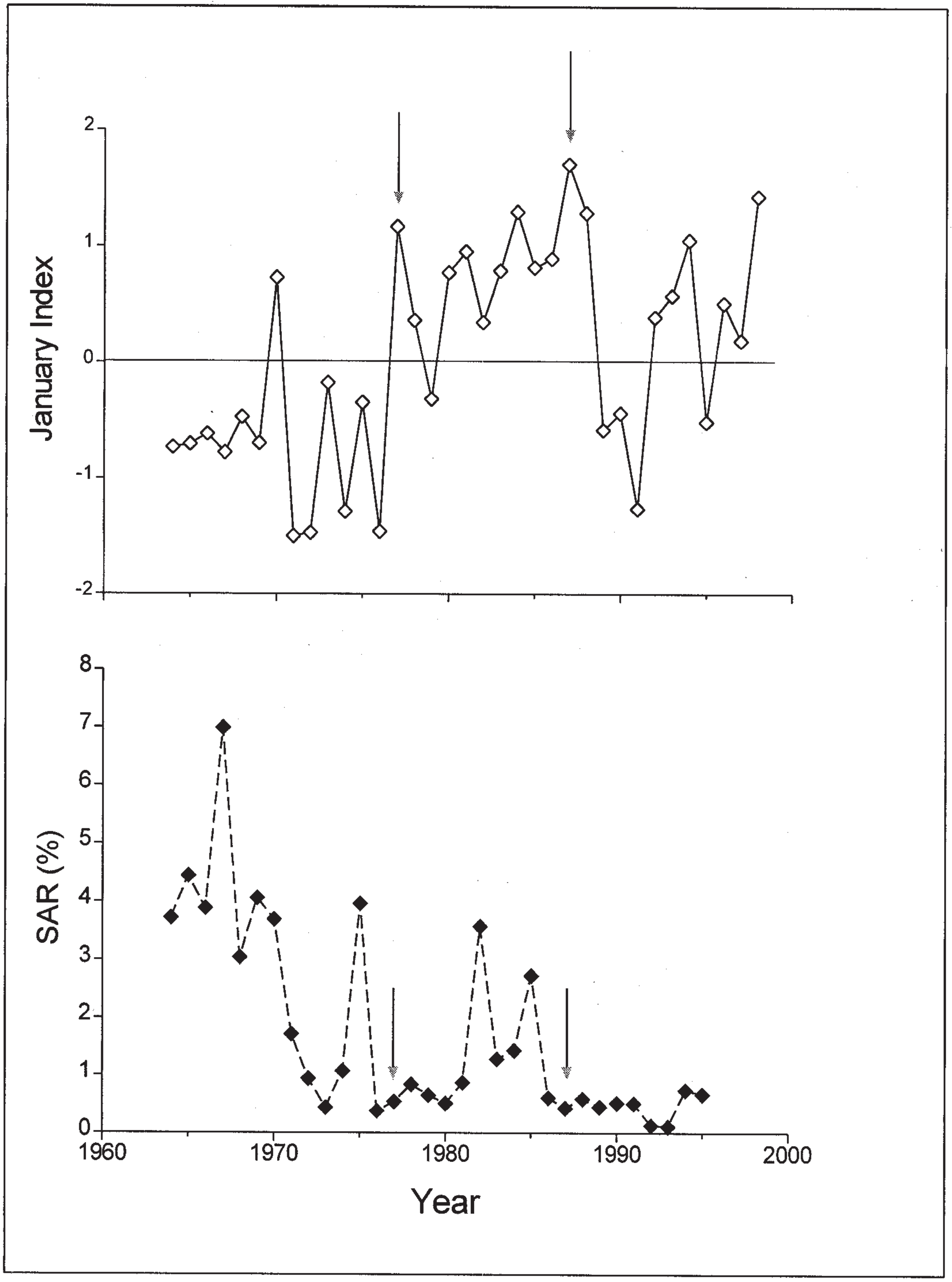

Figure 4.4.1-4: Coincidence in time of anomalies in the Pacific Decadal Oscillation Index and the onset of low smolt-to-adult return rates (SARs) for wild Snake River spring/summer chinook salmon (Williams et al. 1998b). The PDO is a composite index of climatic variation that incorporates the average annual coastal temperature, the average annual basin temperature, and snow depth in March. Arrows indicate two years when high values of the PDO coincided with low SARs. Estimates of the PDO index through March 1988 were received January 20, 1999, from N. Mantua at the internet site: ftp://ftp.atmos.washington.edu/mantua/pnw_impacts/INDICES/PDO.latest. Smolt-to-adult return rates include escapement to the upper-most dam plus harvest.

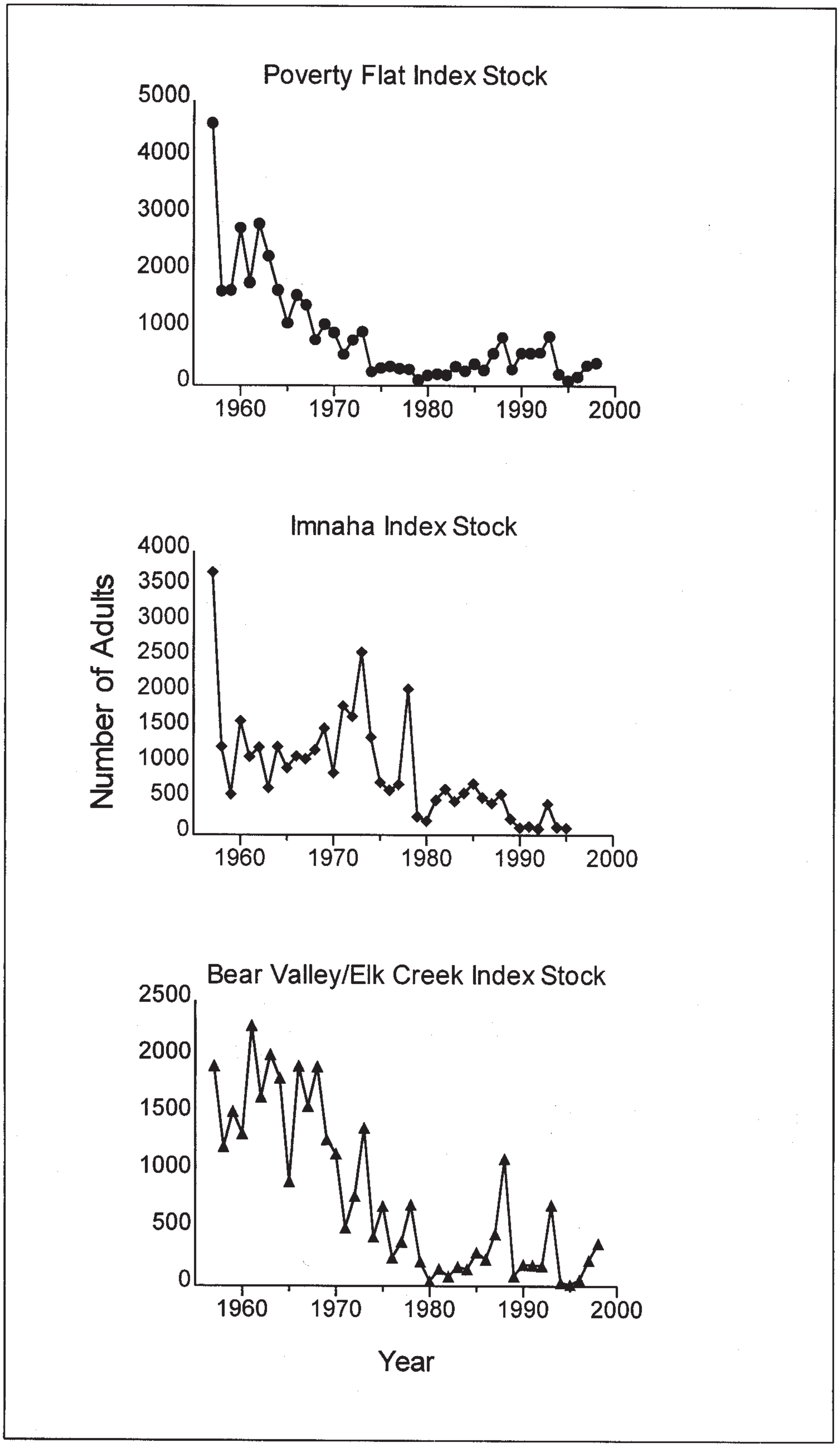

Figure 5.1-1: Declining trends in adult returns for three of the spring/summer chinook index stocks modeled by PATH (Poverty Flat, Imnaha, and Bear Valley/Elk Creek).

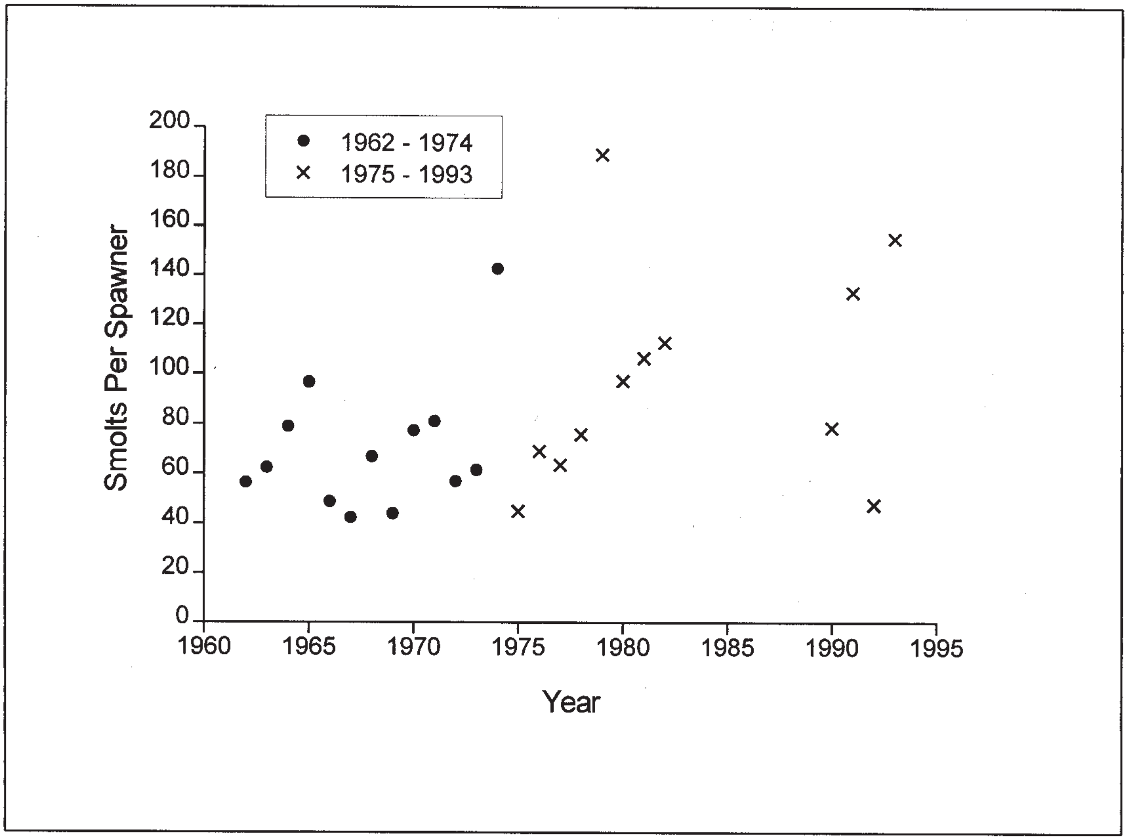

Figure 5.3-1: Number of spring/summer chinook salmon smolts per spawner (collected above Lower Granite Dam; from Petrosky and Schaller 1996). Data are not available for 1983 through 1989. Data from 1962 through 1974 (during the period of construction of the lower Snake River dams) are represented by "·"; data for 1975 through 1993 (after completion of the dams) are represented by "X". Numbers of spawners were calculated by correcting wild escapement for hatchery fish (SP1 estimate method of Petrosky and Schaller 1996). A fish guidance efficiency of 0.56 was assumed for recent estimates of smolt production.

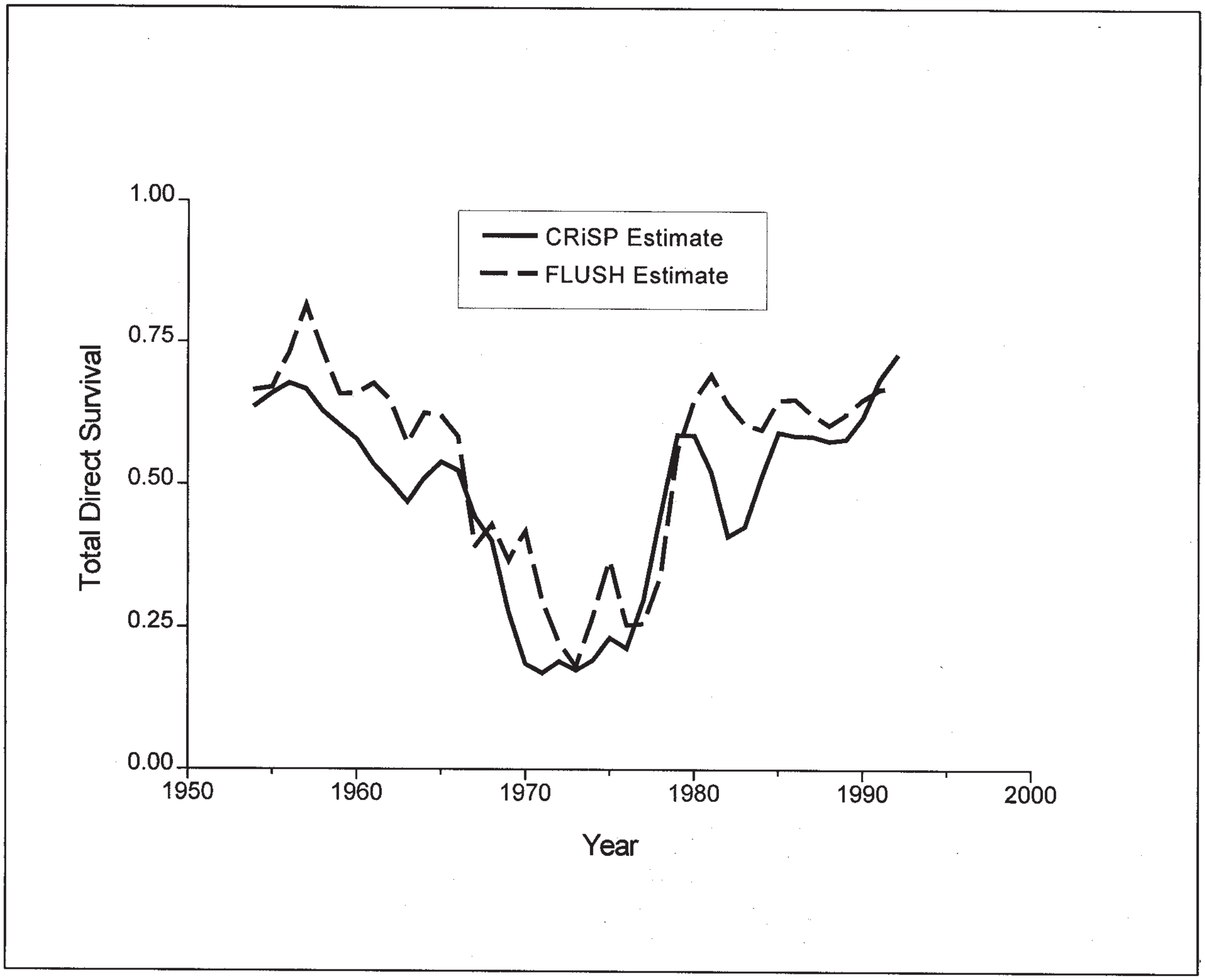

Figure 5.4.1-1: Total direct survival (transported plus in-river migrants) of juvenile spring/summer chinook salmon to below Bonneville Dam, graphed as 5-year moving averages. Note: direct survival does not account for any delayed mortality of either transported or in-river migrants.

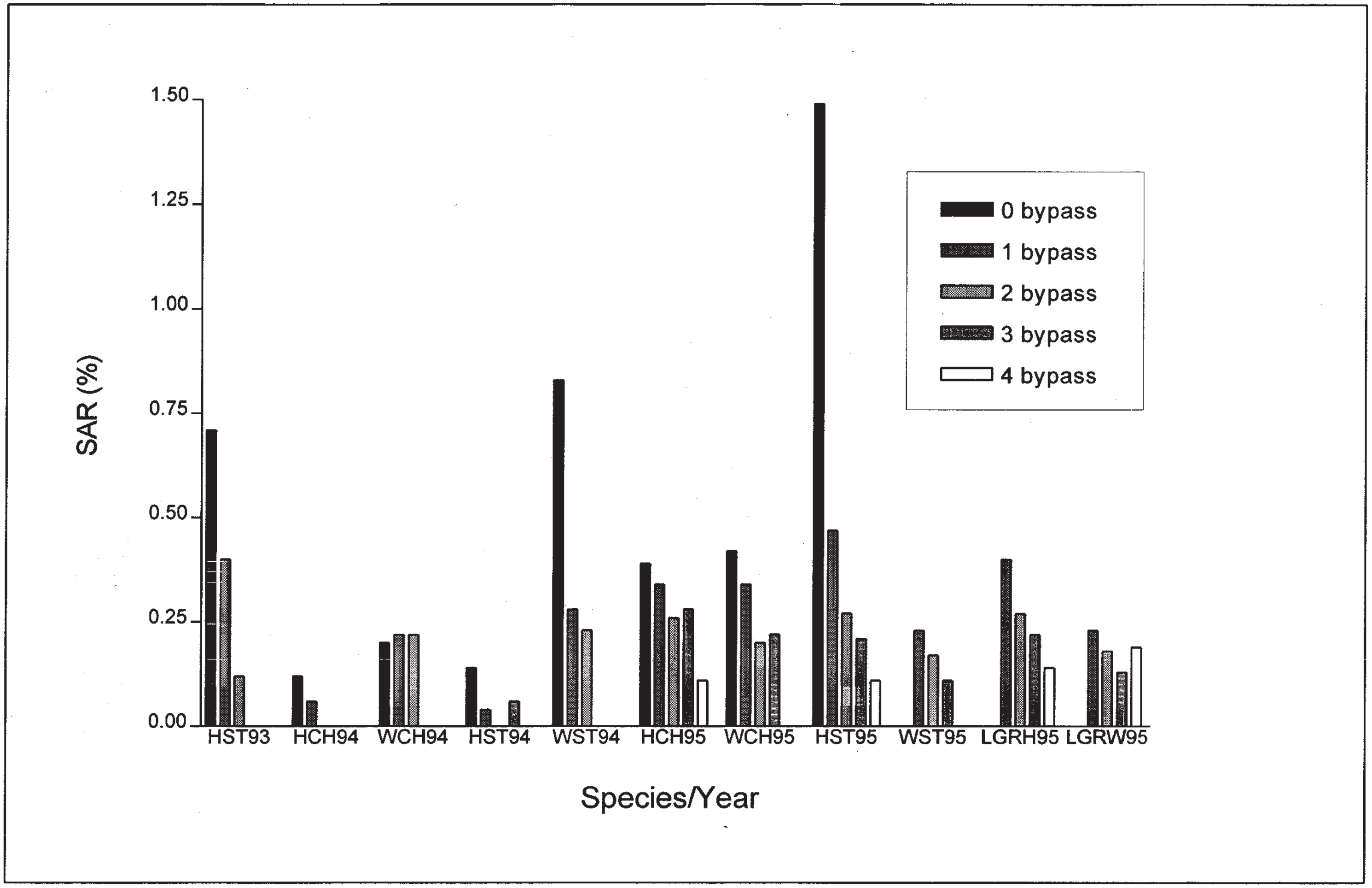

Figure 5.4.3.2-1: Estimated smolt-to-adult return rates (percent) depending on the number of projects at which a juvenile fish was detected in the bypass system during the outmigration. HST-hatchery steelhead; HCH-hatchery chinook salmon; WST-wild steelhead; WCH-wild chinook salmon (all four groups tagged above Lower Granite Dam); LGRH-hatchery chinook salmon tagged at Lower Granite Dam; LGRW-wild chinook salmon tagged at Lower Granite Dam. Numbers identify outmigration year for each group.

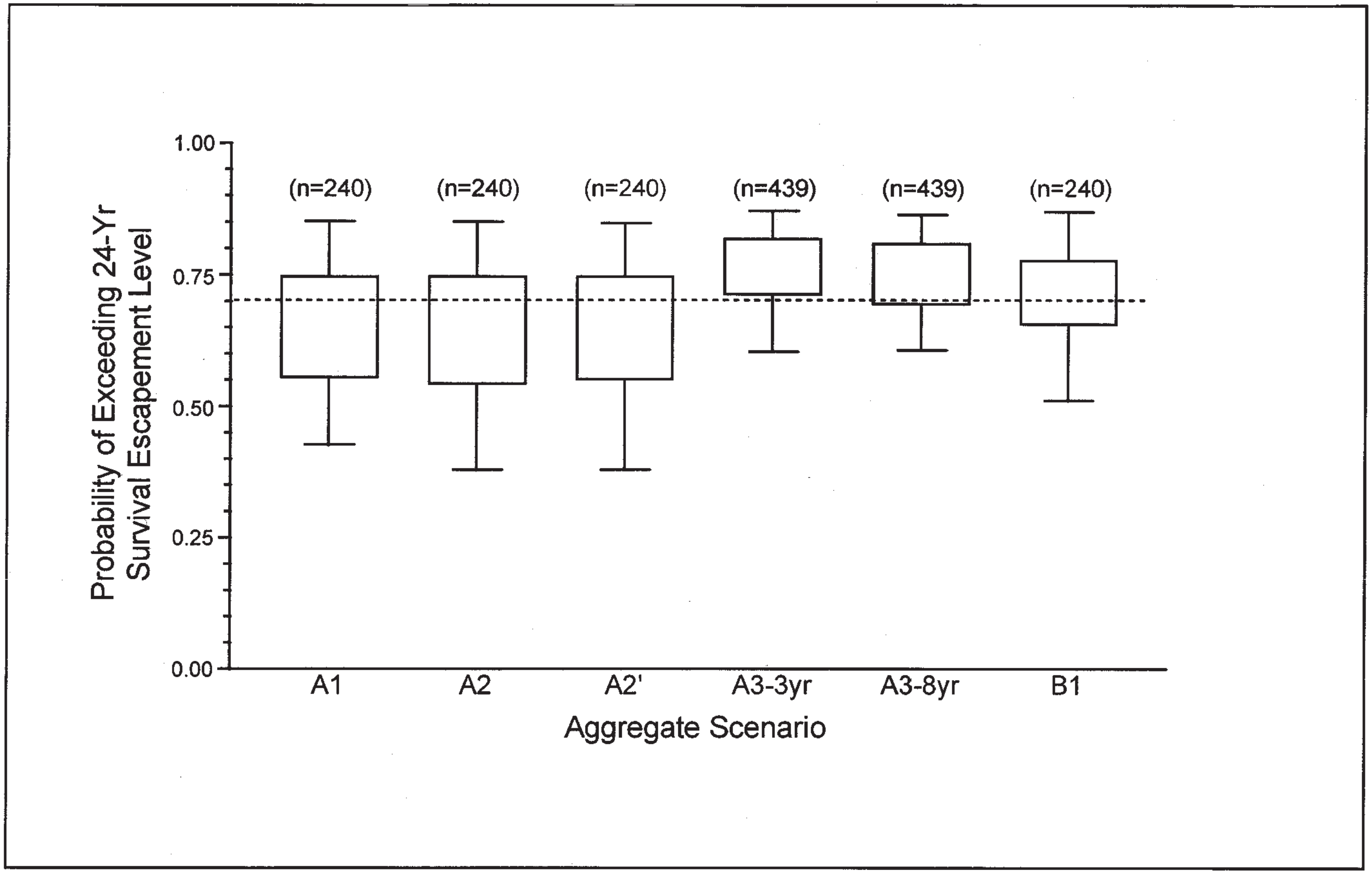

Figure 5.5.1.3-1: Relative probability of exceeding the 24-year survival escapement level for spring/summer chinook salmon under alternatives A1, A2, A2', A3, and B1, according to the PATH prospective life-cycle model. Alternative A3 (drawdown) was evaluated assuming both 3-year and 8-year delays. "n" indicates the number of assumption sets for each scenario. Dashed line indicates the 24-year survival criterion. See text for explanation of "Box and Whisker" plots.

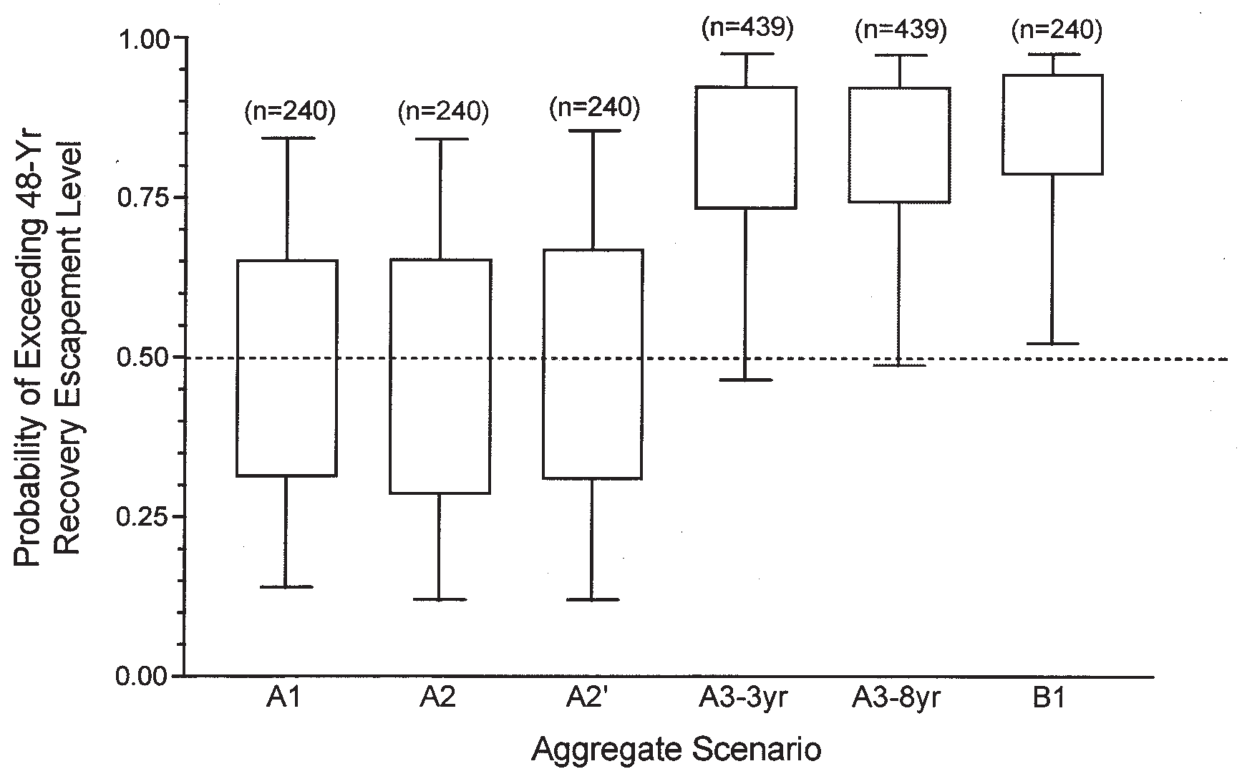

Figure 5.5.1.3-2: Relative probability of exceeding the 48-year recovery escapement level for spring/summer chinook salmon under alternatives A1, A2, A2', A3, and B1, according to the PATH prospective life-cycle model. Alternative A3 (drawdown) was evaluated assuming both 3-year and 8-year delays. "n" indicates the number of assumption sets for each scenario. Dashed line indicates the 48-year recovery criterion. See text for explanation of "Box and Whisker" plots.

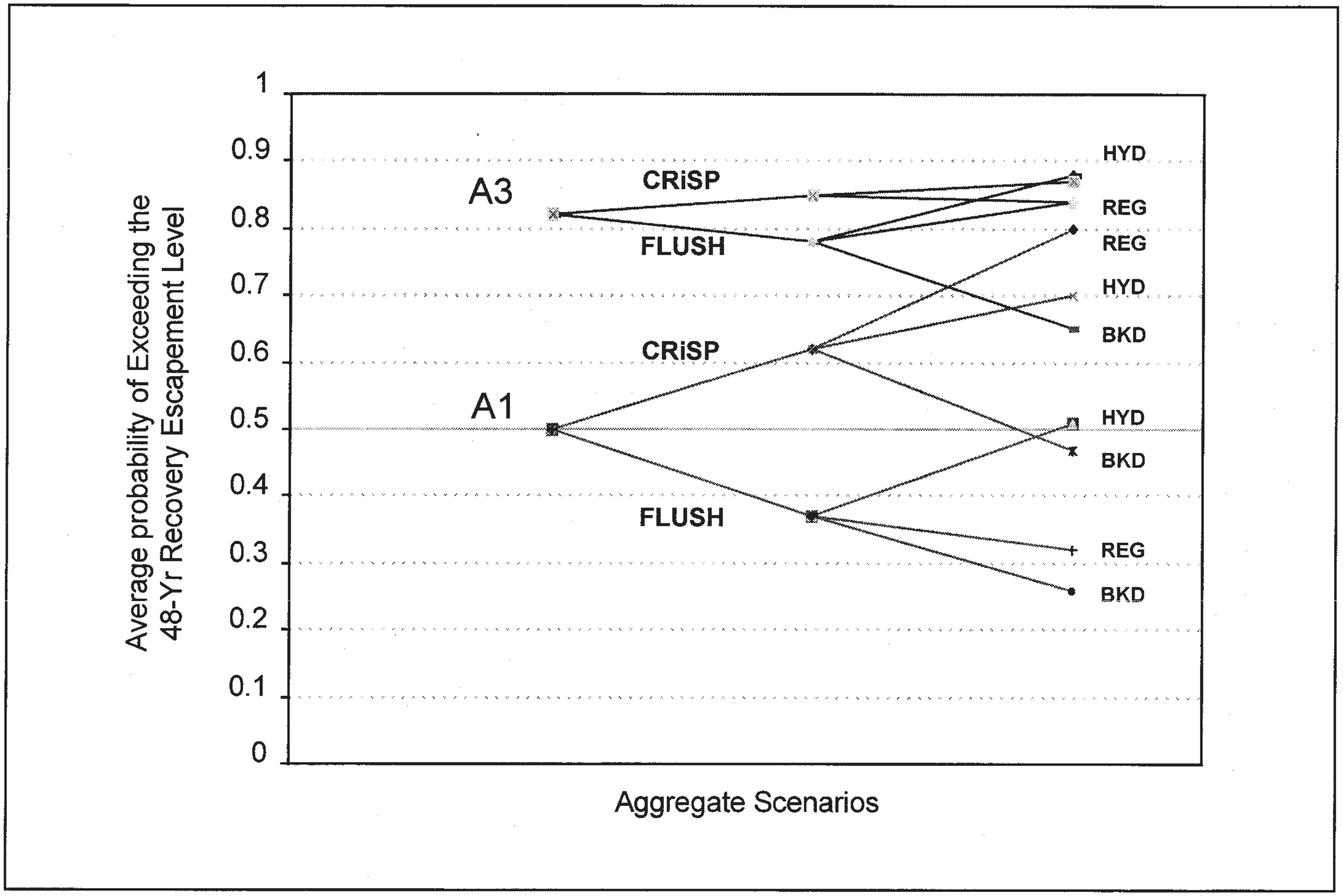

Figure 5.5.1.4-1: Relationship between different combinations of assumptions and the average probability of exceeding the 48-year recovery escapement level, as predicted by the PATH life-cycle model. Solid horizontal line indicates the 48-year recovery criterion.

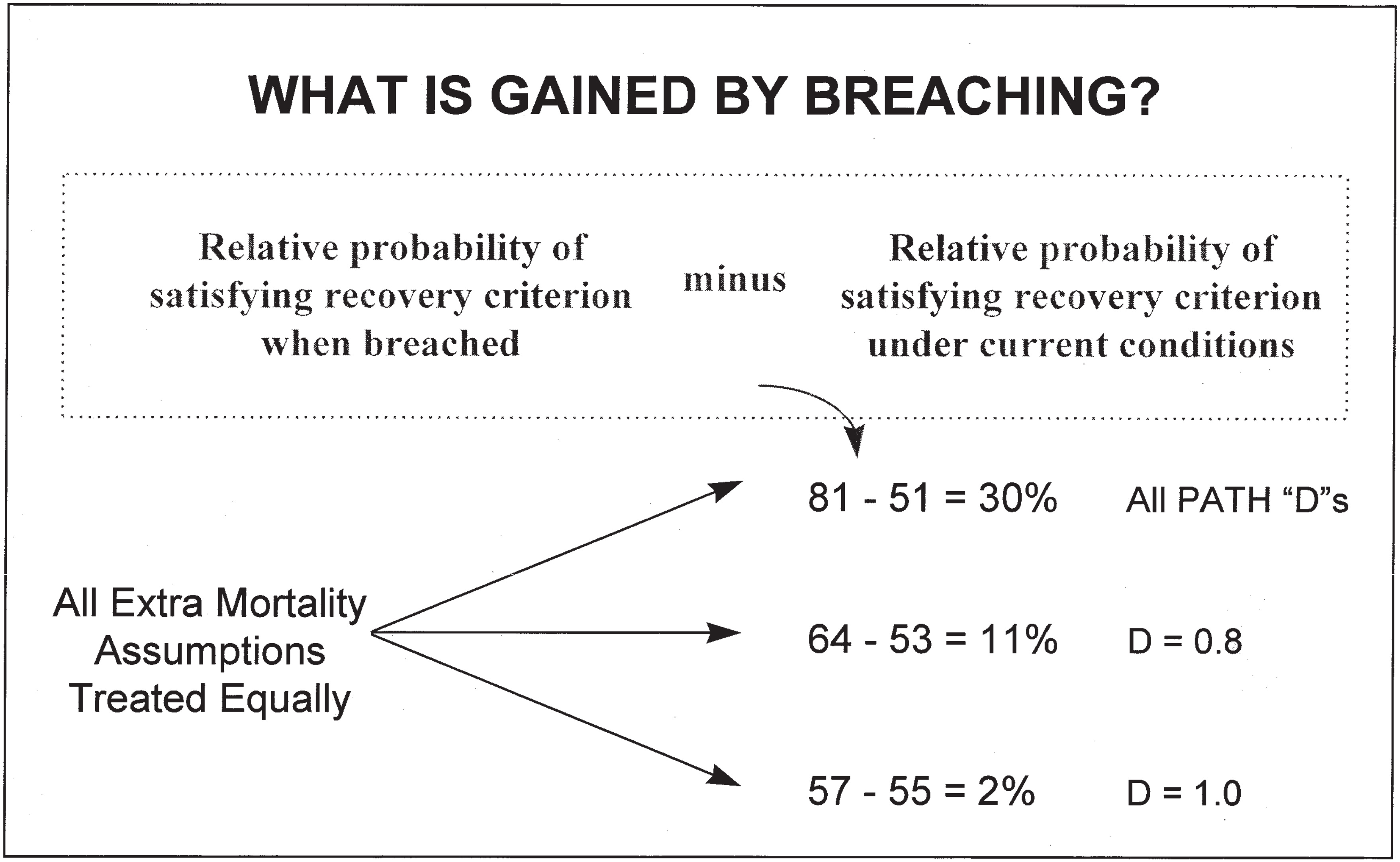

Figure 5.5.1.4-2: Demonstration of the increase in the relative probability of exceeding the 48-year recovery escapement level under breaching (A3) compared to the current condition (A1), shown for three alternative values of D.

If $D = 0.8$,

the increased probability of meeting the recovery threshold under breaching is very sensitive to assumptions about extra mortality.

Assumed Source of Extra Mortality	Increased Probability of Meeting Recovery Goal (under breaching)
Hydrosystem	19%
Degraded Stock Viability	6%
Ocean Regime Shift	2%

Figure 5.5.1.4-3: Sensitivity of the increased probability of meeting the 48-year recovery goal (i.e., 48-year recovery escapement level) under breaching to assumptions about the source of extra mortality.

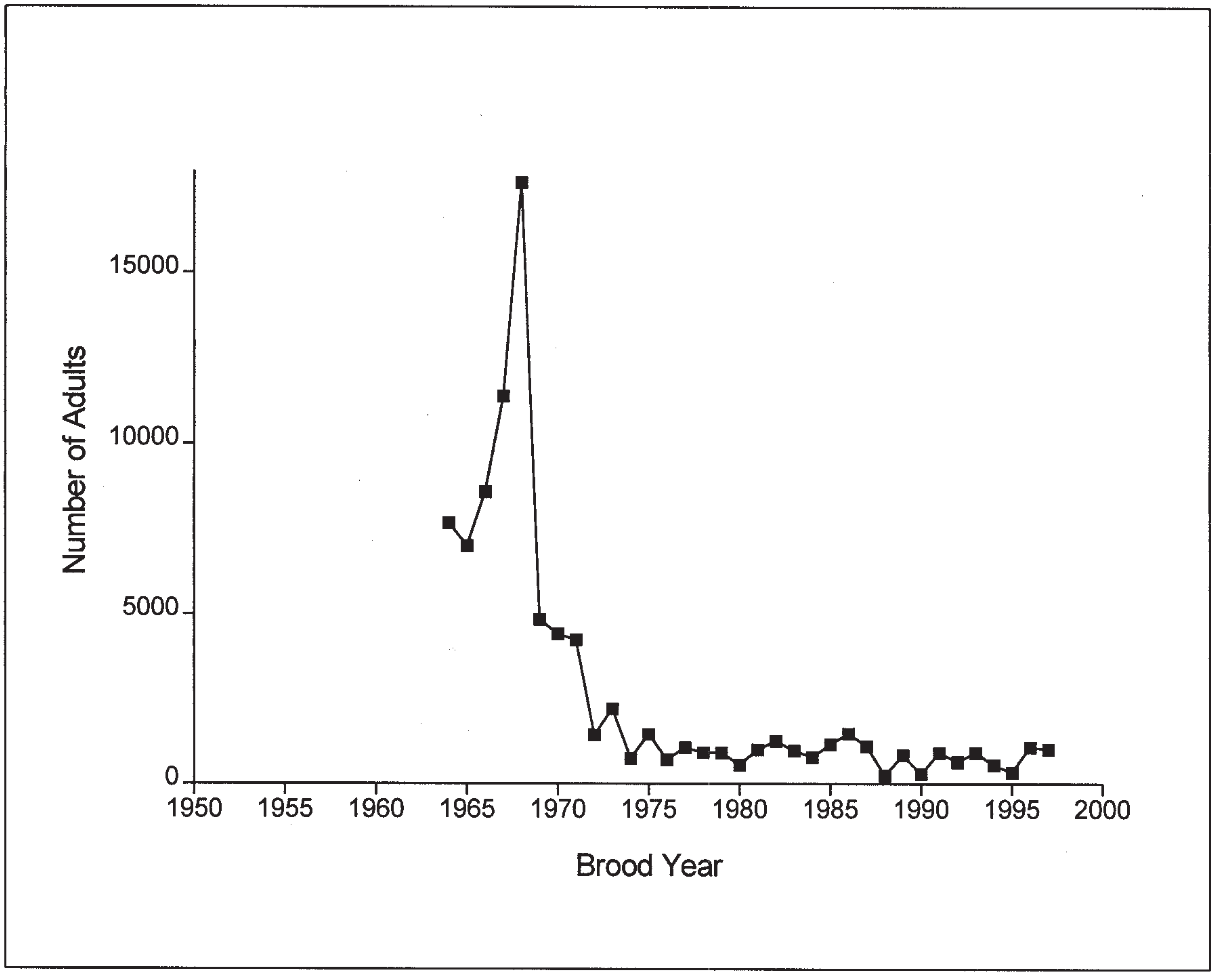

Figure 6.1.1-1: Wild fall chinook salmon spawner abundance (count at uppermost Snake River dam) from run reconstructions in Marmorek et al. (1998).

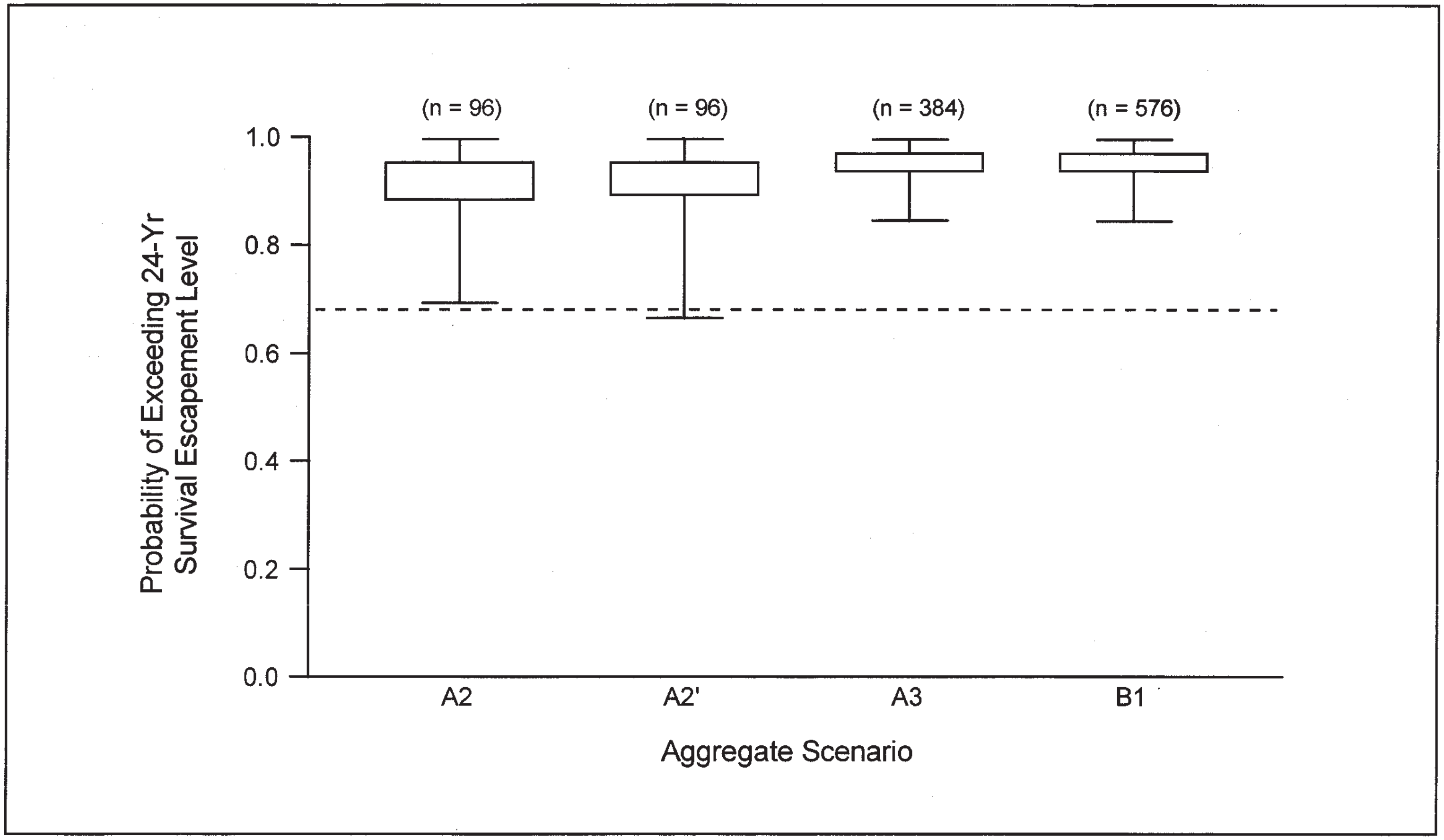

Figure 6.5.1-1: Relative probability of exceeding the 24-year survival escapement level for fall chinook salmon under alternatives A2, A2', A3, and B1, according to the PATH life-cycle model. "n" indicates the number of assumption sets for each scenario. Dashed line indicates the 24-year survival criterion. See text for explanation of "Box and Whisker" plots.

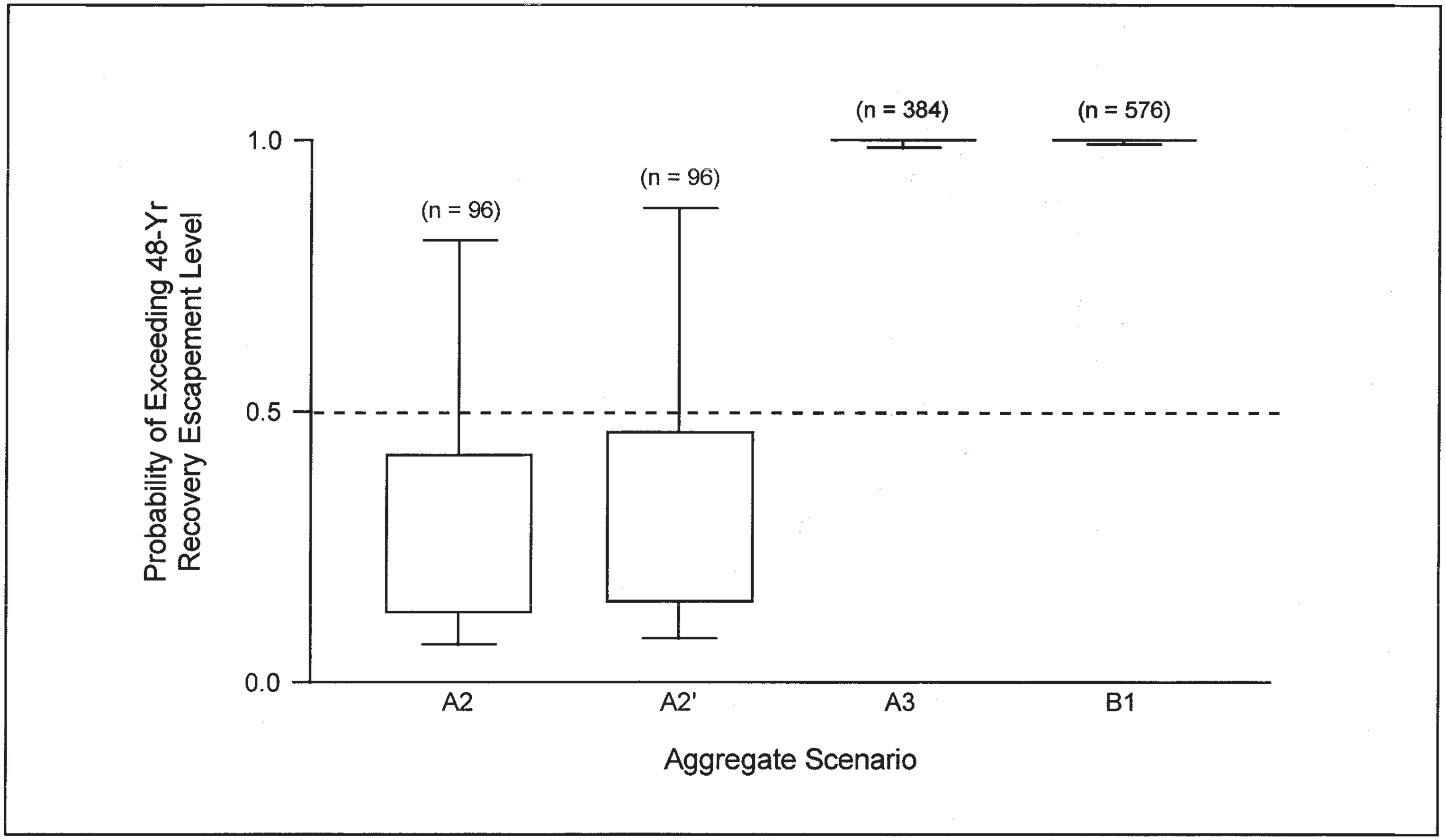

Figure 6.5.1-2: Relative probability of exceeding the 48-year recovery escapement level for fall chinook salmon under alternatives A2, A2', A3, and B1, according to the PATH life-cycle model. "n" indicates the number of assumption sets for each scenario. Dashed line indicates the 48-year recovery criterion. See text for explanation of "Box and Whisker" plots.

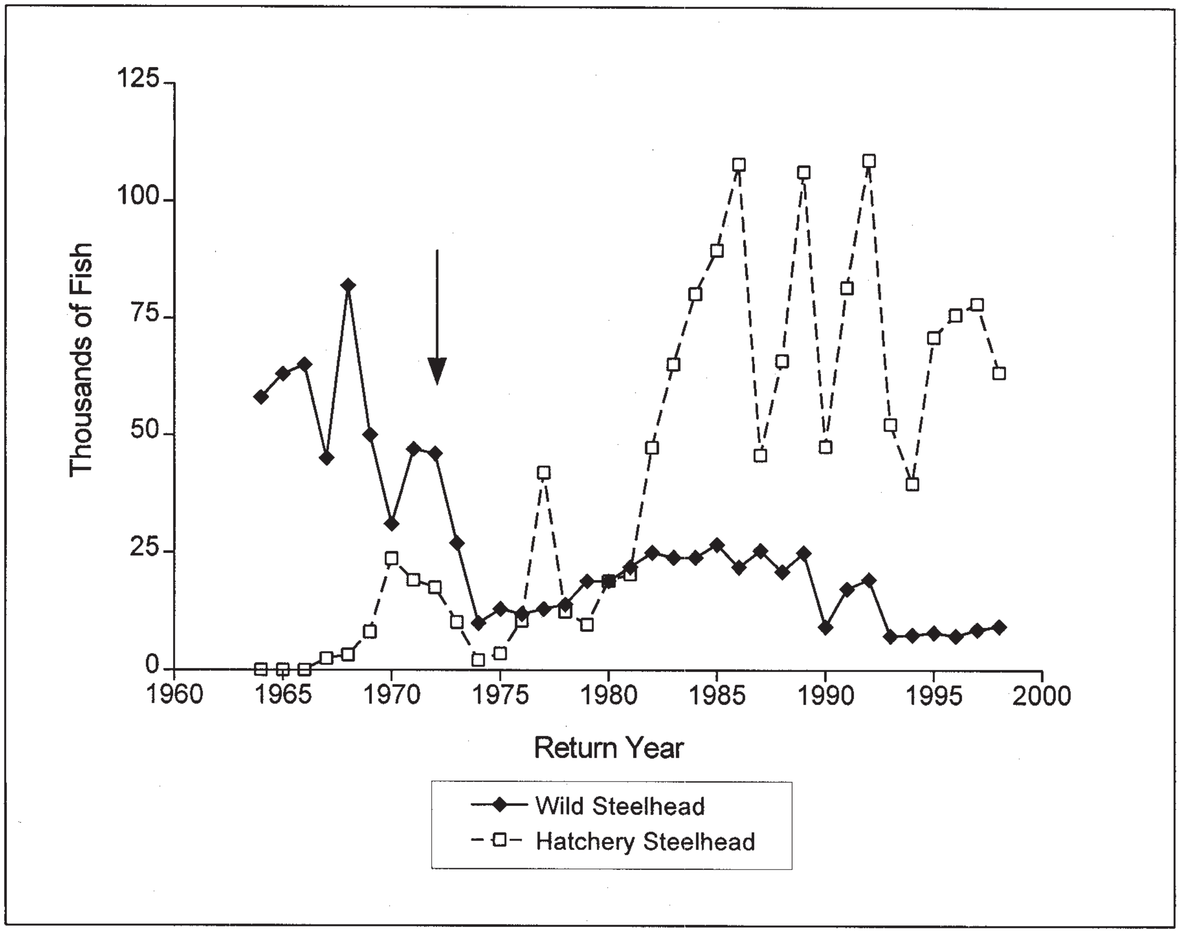

Figure 7.1-1: Estimated returns of adult wild and hatchery steelhead to the uppermost dam on the lower Snake River. Uppermost dams were: Ice Harbor from 1964 through 1968; Lower Monumental during 1969; Little Goose from 1970 through 1974; and Lower Granite in all subsequent years. Arrow represents construction of Dworshak Dam, which blocked access to the North Fork Clearwater River, a significant B-run steelhead spawning area. Reproduced from TAC (1997). For comparison with other figures, "return year" is, on average, "migration year" + 1 for Snake River steelhead.

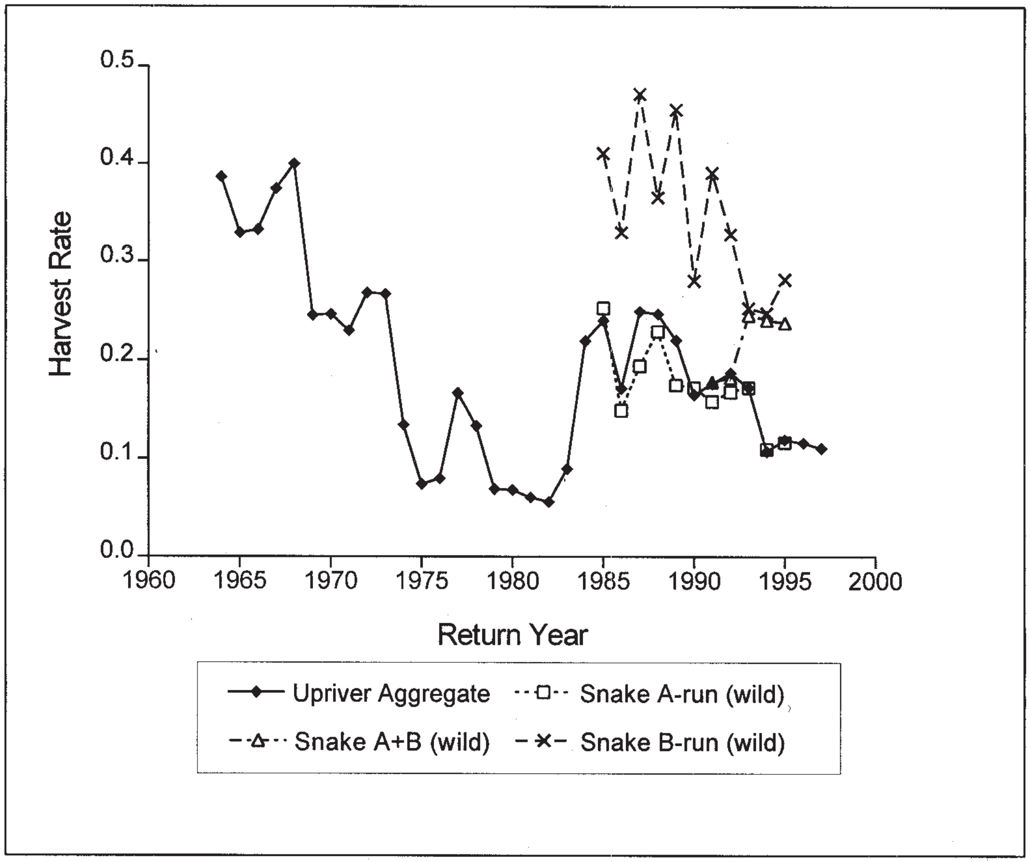

Figure 7.2.1-1: Harvest rates for Columbia River Basin steelhead. Harvest of the "Upriver Steelhead Aggregate" is calculated as the combined Zone 1-5 fishery divided by the minimum run size during 1964 through 1997 (ODFW and WDFW 1998). "Snake A-Run (Wild)" is for mainstem harvest of wild A-run steelhead, estimated using the length method described in TAC (1997). "Snake B-Run (Wild)" is calculated in the same manner (TAC 1997). Harvest of the "Snake A+B (Wild)" combines the catches of wild Snake River steelhead above-Bonneville and above-McNary dams, divided by the reconstructed run size for that group at Bonneville Dam (TAC 1998 and Marmorek et al. 1998). For comparison with other figures, "return year" is, on average, "migration year" + 1 for Snake River steelhead.

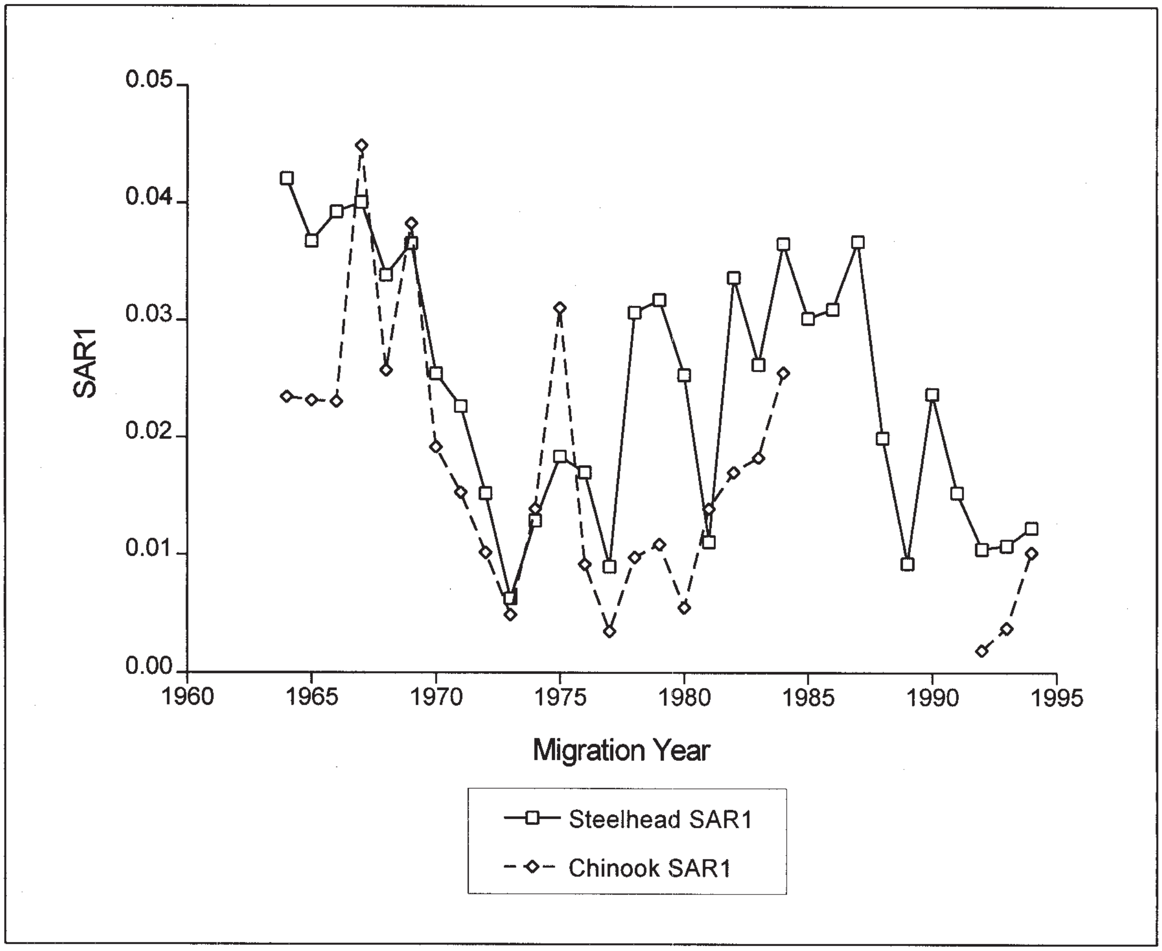

Figure 7.4-1: Estimates of Escapement SAR (to upper dam) for Snake River steelhead (Steelhead SAR1). Spring/summer chinook salmon Escapement SAR (Chinook SAR1) is displayed for comparison. Estimates from Petrosky (1998) and Petrosky and Schaller (1998).

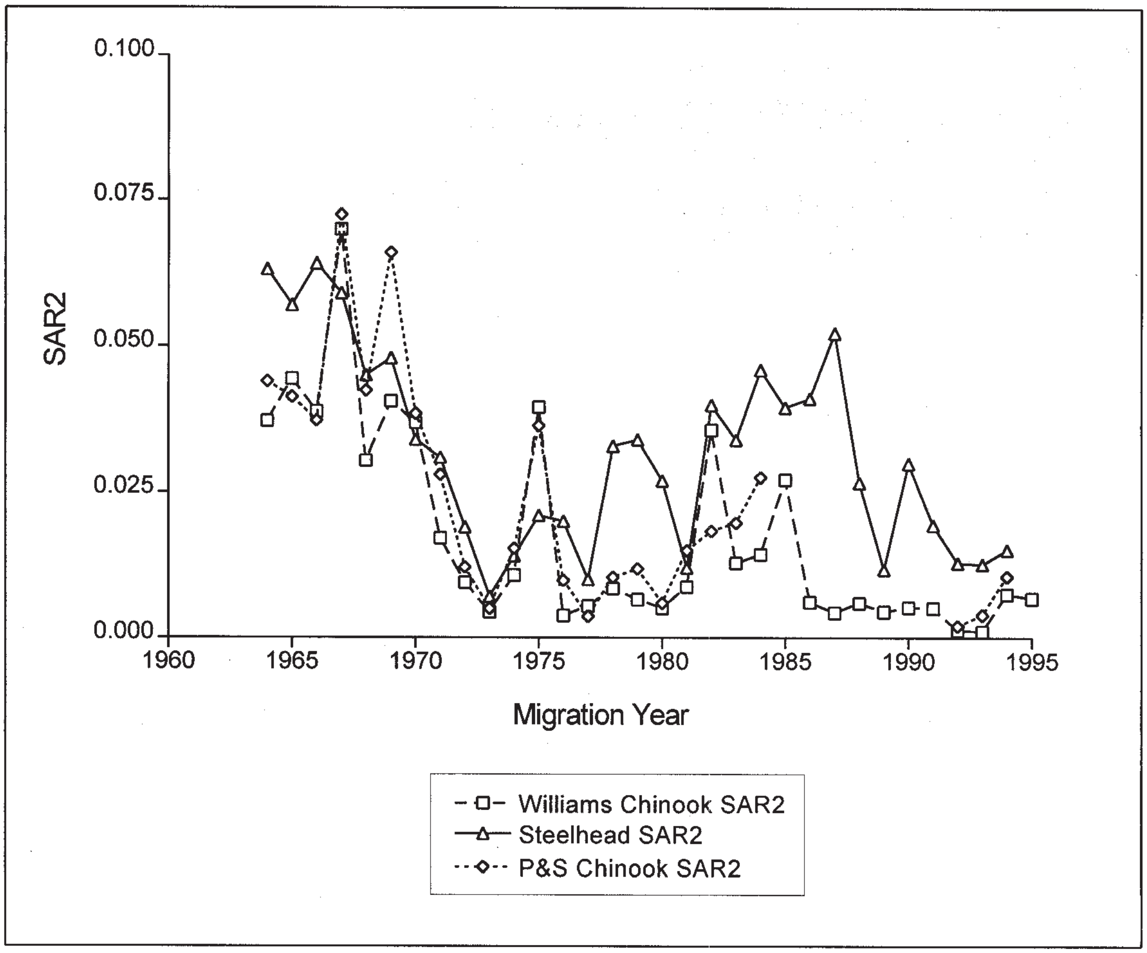

Figure 7.4-2: Estimates of Escapement + Harvest SAR (which counts harvested fish towards escapement) for Snake River steelhead (Steelhead SAR2). Two estimates of spring/summer chinook salmon Escapement + Harvest SAR are displayed for comparison. Estimates for steelhead SARs are from Petrosky (1998); estimates for chinook salmon SARs from Petrosky and Schaller (1998) and Williams et al. (1998).

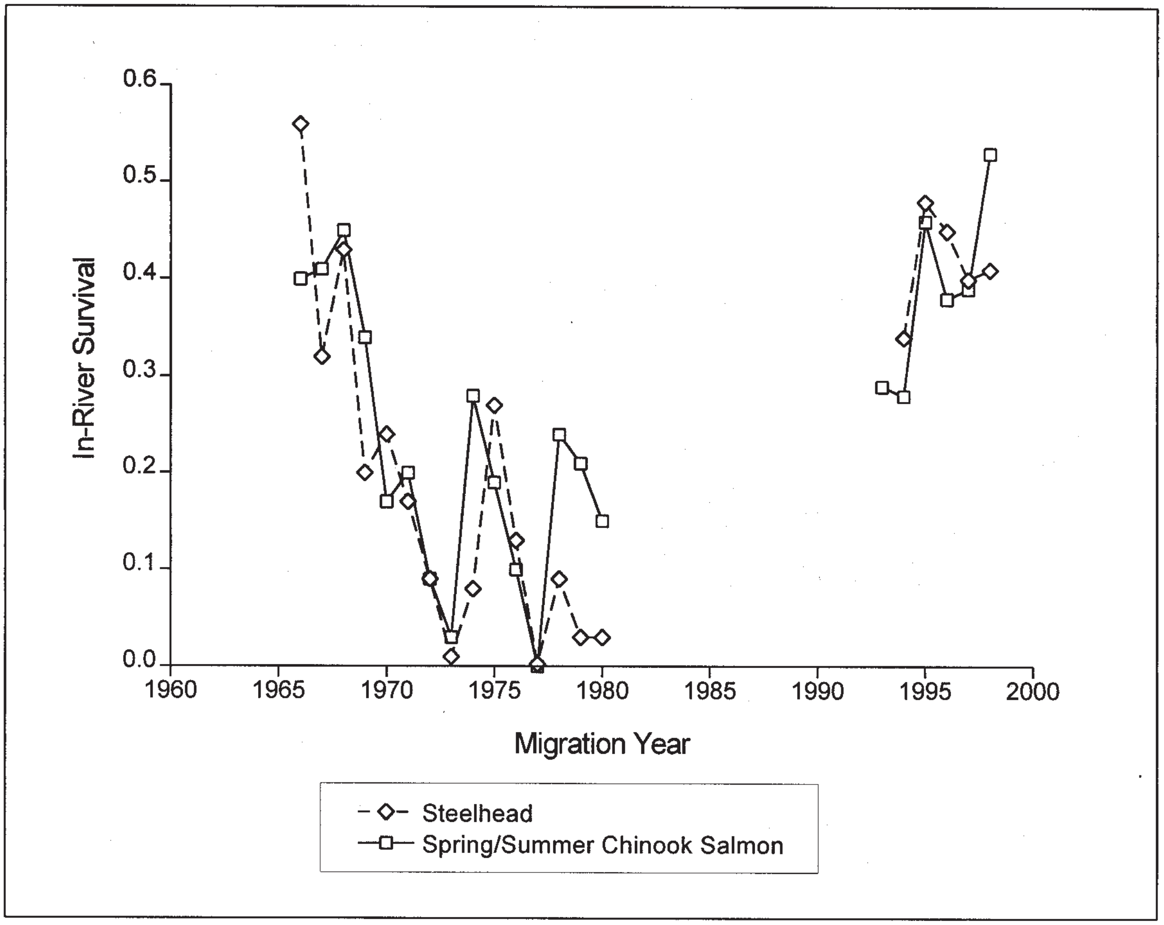

Figure 7.4.1-1: NMFS reach survival estimates, expanded to represent survival through all lower Snake River and lower Columbia River projects in existence during a particular period (1966 through 1967 = 4 dams; 1968 = 5 dams; 1969 = 6 dams; 1970 through 1974 = 7 dams; 1975 through 1997 = 8 dams) using the method in Smith and Williams (1999). Estimates for 1994 through 1997 are multiplied by 0.9 to approximate the overestimation expended because of the different in-river passage experience of PIT-tagged fish compared to the experience of fish in the run-at-large (see text). Note that the in-river survival estimates for spring/summer chinook produced by this method differ from the estimates produced by detailed passage models and are displayed only to allow direct comparison with steelhead estimates.

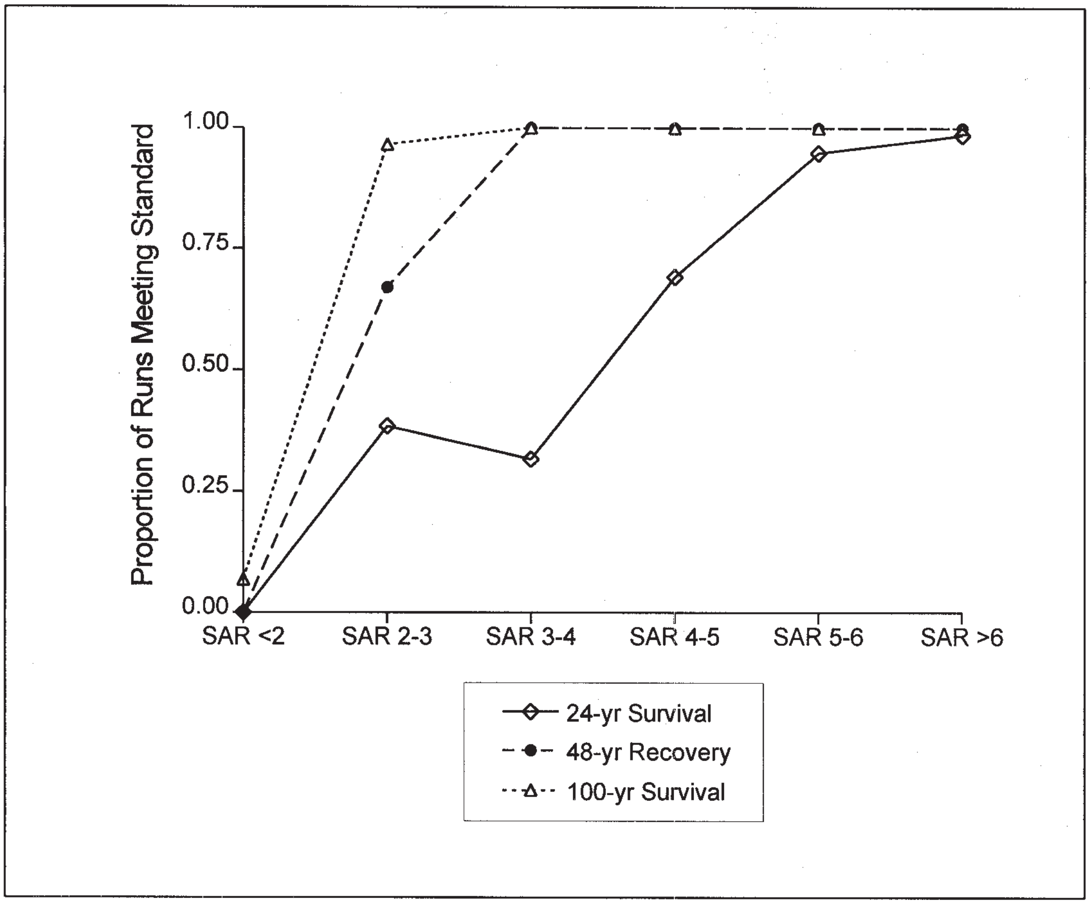

Figure 7.5-1: Probability that model runs resulting in 100-year median escapement SAR (generated by PATH life-cycle model as SAR to the upper dam) meet survival and recovery criteria for Snake River spring/summer chinook salmon. For example, for model runs resulting in a simulated median escapement SAR between 3.0 and 3.99, slightly more than 30% of these runs meet the 24-year survival criterion, slightly less than 70% meet the 48-year recovery criterion, and all of them meet the 100-year survival criterion. Certainty of meeting the 100-year survival criterion requires a median escapement SAR of at least 3%, certainty of meeting the 48-year recovery criterion requires a median escapement SAR of at least 4%, and certainty of meeting the 24-year survival criterion requires a median escapement SAR greater than 6%.

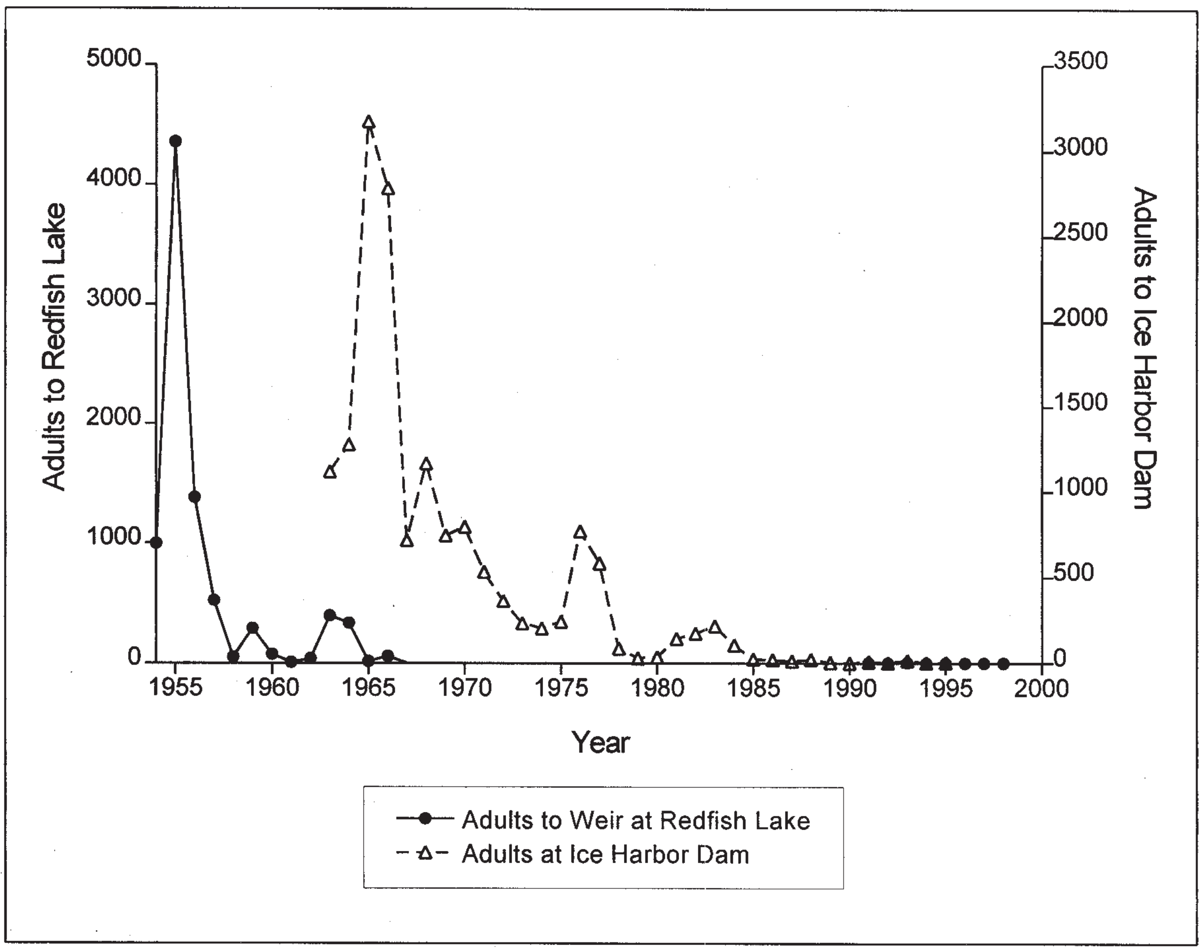

Figure 8.1-1: Escapement of Snake River sockeye salmon to the weir at the outlet from Redfish Lake and to Ice Harbor Dam. Counts at Redfish Lake from Kiefer et al. (1991). Counts at Ice Harbor Dam from ODFW and WDFW (1998).

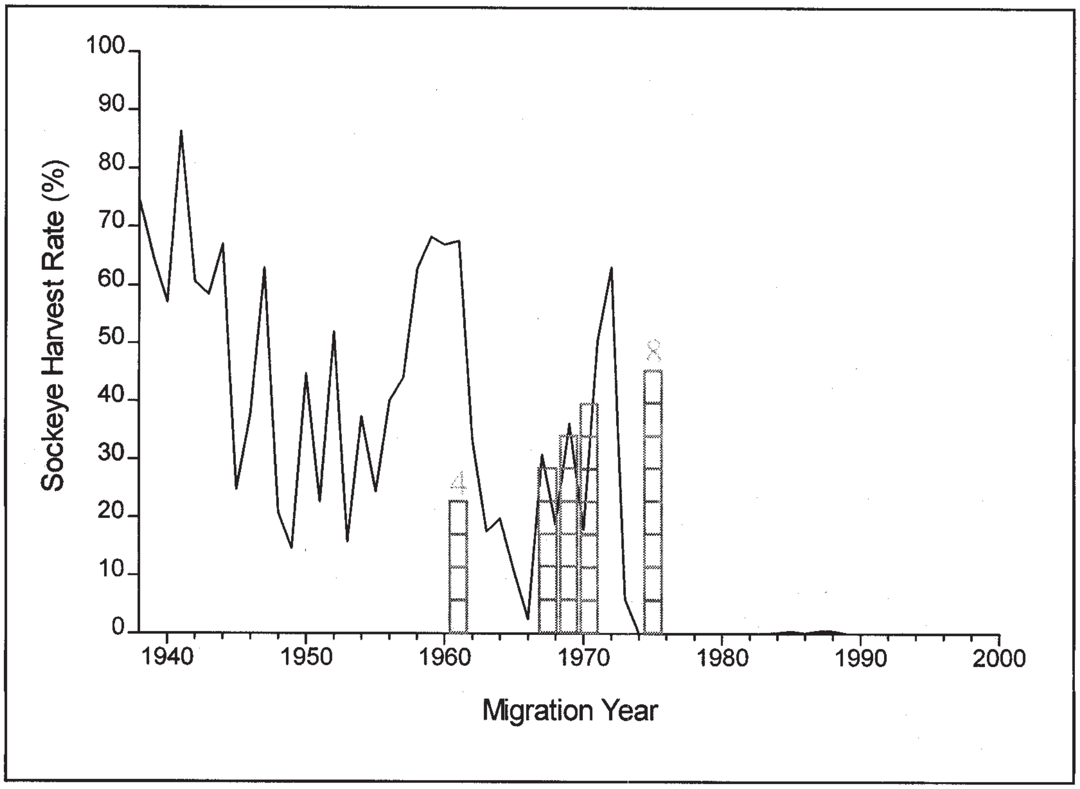

Figure 8.2.1-1: Mainstem harvest rates for Snake River sockeye salmon in Zones 1 through 6 and cumulative number of mainstem (lower Snake and lower Columbia rivers) dams. Harvest rates calculated as the proportion of the run to Bonneville Dam (ODFW and WDFW 1998).

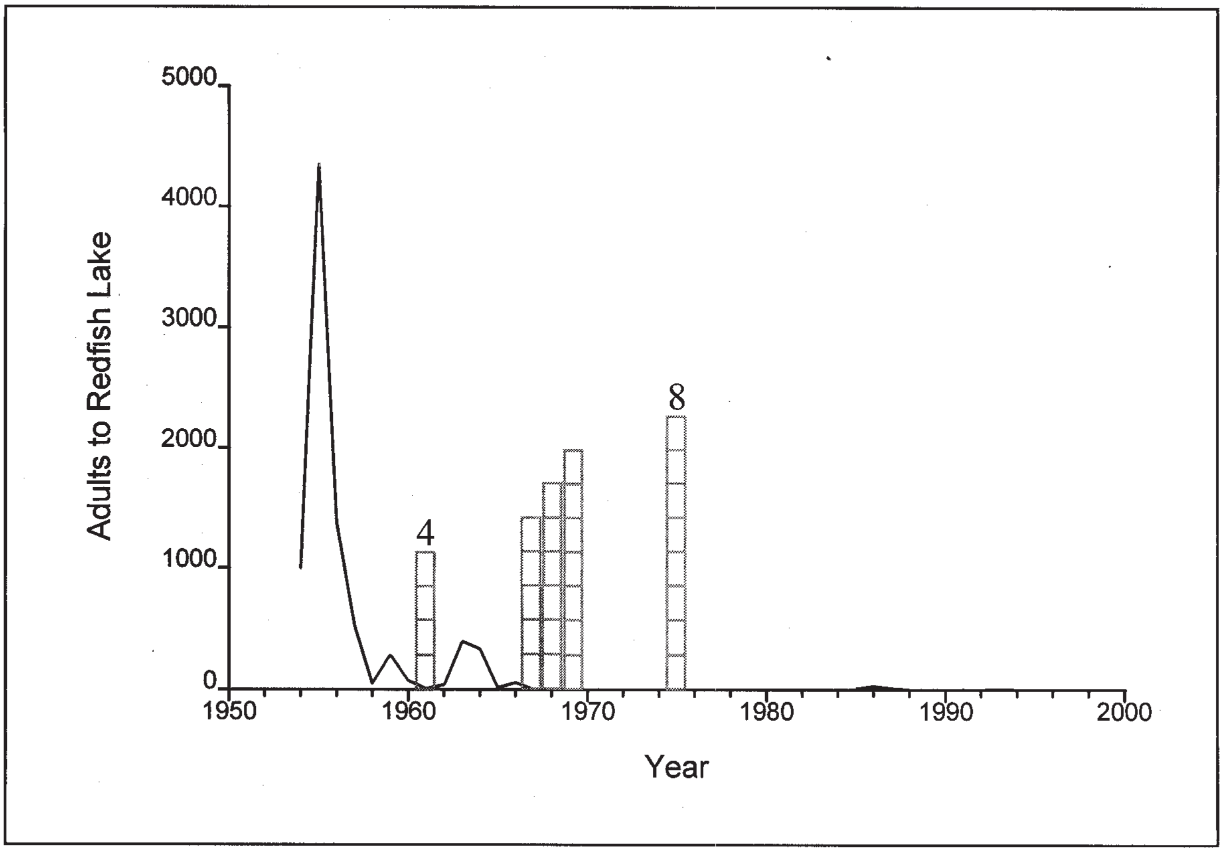

Figure 8.2.2.1-1: Escapement of Snake River sockeye salmon to the weir at the outlet from Redfish Lake (Kiefer et al. 1991) and cumulative number of mainstem (lower Snake and lower Columbia rivers) dams.

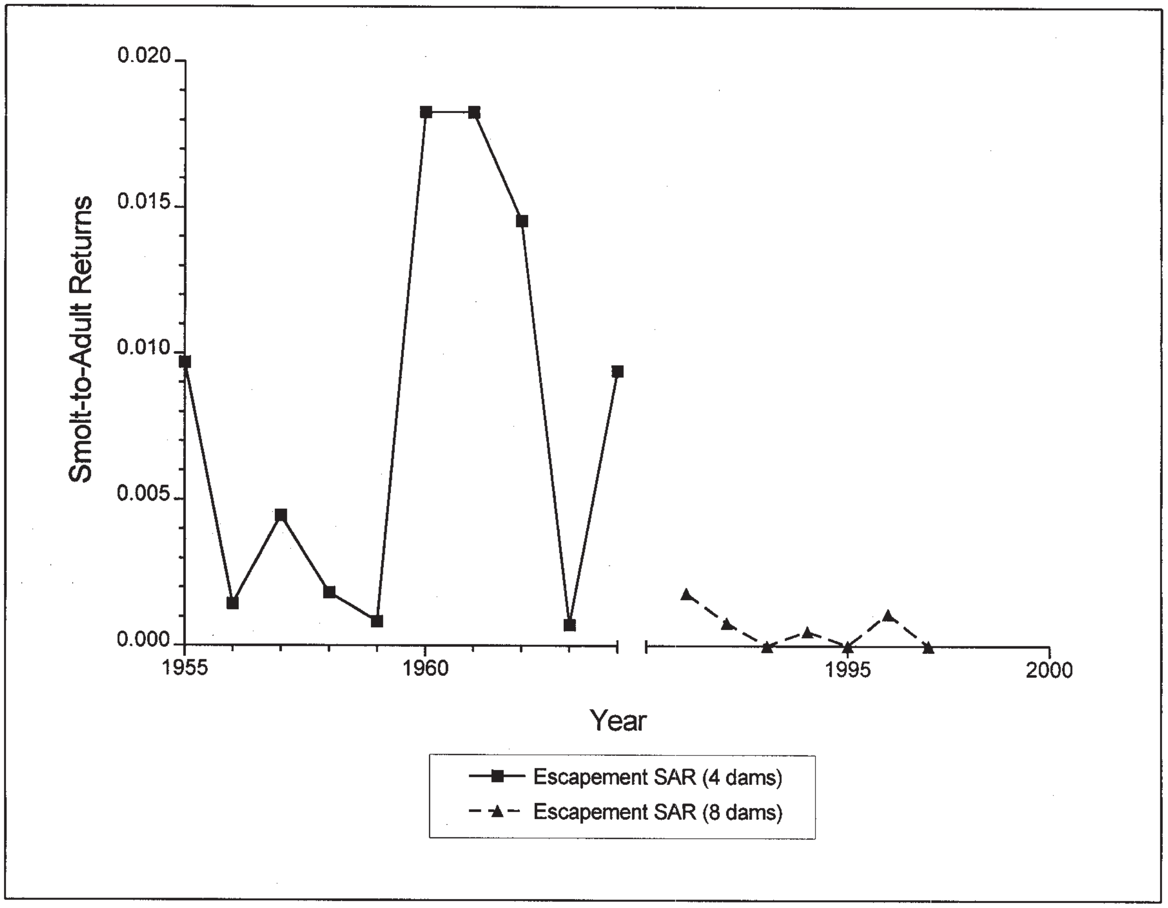

Figure 8.4-1: Smolt-to-adult return rates for Snake River sockeye salmon during periods when up to four and between five and eight mainstem dams were in place, respectively. SARs include escapement (but not harvest) (Bjornn et al. 1968; P. Kline, pers. comm. -- as described in the text).

ATTACHMENT

PATH Snake River Spring/Summer Chinook Models

Delta Model Description

The Delta model is described in Wilson et al. (1997), Marmorek et al. (1998a, p. A87-A91), Deriso (1997), and Marmorek et al. (1998b). The mathematical representation is:

$$\ln(R_{t,i}) = (1+p)\ln(S_{t,i}) + a_i - b_i S_{t,i} - M_{t,i} - \Delta m_{t,i} + \delta_t + \varepsilon_{t,i}$$

The terms in this equation and their derivations differ between the retrospective and the prospective implementations of the Delta model.

$R_{t,i}$ = Adult returns to the Columbia River mouth (recruitment) originating from spawning in year t and river sub-basin i.

Retrospective Implementation: Estimates of Columbia River recruits from Beamesderfer et al. (1997) are **input** to the retrospective model.

Prospective Implementation: Columbia River recruits are **estimated** by the prospective model from all other terms in the equation.

$S_{t,i}$ = Spawners in year t and river sub-basin i.

Retrospective Implementation: Estimates of spawners from Beamesderfer et al. (1997) are **input** to the retrospective model.

Prospective Implementation: In the first few years of the prospective simulation, available estimates of spawner abundance are **input** to the prospective model, as in the retrospective implementation. For subsequent years, the number of spawners is **estimated** by the prospective model as:

$$S_{t,i} = \Sigma_a f_{t,a,i}\, s_{t,i}\, R_{t-a,I}$$

in which a represents age and a fraction $f_{t,a,i}$ of total recruitment $R_{t-a,i}$ produced in brood year t-a returns in year t and experiences up-river survival to the spawning ground of $s_{t,i}$. The previous brood years' recruitment is **estimated** within the prospective model, as described above. The other terms require **input** to the prospective model of: (1) a prospective conversion factor from Bonneville Dam through Lower Granite Dam, which accounts for all non-fishery related losses during up-river passage; (2) an age-specific exploitation fraction, which is the total loss due to in-river fisheries; and (3) pre-spawning mortality, which represents loss of adults between Lower Granite Dam and the spawning grounds. A stock-specific maturity schedule, selected at random from the brood year 1963-1993 estimates (H. Schaller, ODFW, pers. comm. to R. Deriso) was applied in the prospective analysis. Details are included in Deriso (1997).

$a_{,i}$ = Ricker *a* parameter, which represents inherent stock productivity and depends on sub-basin *i*.

Retrospective Implementation: This parameter is **estimated** by the retrospective modeling procedure. The result is a posterior probability distribution of estimates.

Prospective Implementation: This parameter is **input** to the prospective model. Estimates are drawn at random from the posterior probability distribution generated by the retrospective model. One modification of this implementation involves input of a proportional change scalar by which the retrospective Ricker *a* parameter selected for each simulation is multiplied, for use in habitat sensitivity analyses.

b_i = Ricker b parameter, which represents stock carrying capacity and depends on sub-basin *i*.

Retrospective Implementation: This parameter is **estimated** by the retrospective modeling procedure. The result is a posterior probability distribution of estimates.

Prospective Implementation: This parameter is **input** to the prospective model. Estimates are drawn at random from the posterior probability distribution generated by the retrospective model.

p = depensation parameter, which represents a decline in the number of recruits per spawner as spawner abundance declines and which is applied to all stocks.

Retrospective Implementation: This parameter is **estimated** by the retrospective modeling procedure. The result is a posterior probability distribution of estimates.

Prospective Implementation: This parameter is **input** to the prospective model. Estimates are drawn at random from the posterior probability distribution generated by the retrospective model.

$M_{t,i}$ = direct passage mortality, which depends on year and includes combined mortality of both transported and non-transported smolts. For all sub-basins *i* within the Snake River sub-region, mortality is from the head of Lower Granite pool to below Bonneville Dam.

Retrospective Implementation: This survival rate is **input** to the retrospective model from FLUSH and CRiSP passage model estimates.

Prospective Implementation: This survival rate is combined with the $\Delta m_{t,i}$ term in the prospective implementation, as described for $\Delta m_{t,i}$ below.

$\Delta m_{t,i}$ = extra mortality rate, which depends on year and region. "Extra mortality" is any mortality occurring outside the juvenile migration corridor that is not accounted for by the other terms in this model. That is, it is not accounted for by: (1) productivity parameters in the spawner-recruit relationship (*a*, *b*, and *p*); (2) estimates of direct mortality within the migration corridor ($M_{t,i}$); (3) common year effects influencing both Snake River and Lower Columbia River stocks (δ_t); and (4) random effects specific to each stock in each year, as represented by the $\varepsilon_{t,i}$ term.

Retrospective Implementation: This term is estimated as:

$$\Delta m_{t,i} = m_{t,i} - M_{t,i}$$

with $M_{t,i}$ defined as above and $m_{t,i}$ defined as:

$$m_{t,i} = X * n_{t,i} + \mu_t$$

These terms are defined and discussed in Deriso et al. (1996), Deriso (1977), and Marmorek et al. (1998c). Briefly, $n_{t,i}$ is **input** to the retrospective model and represents the total number of "X-level"dams (defined as Bonneville, John Day, and/or The Dalles) that stock *i* must pass in year *t*. *X* is **estimated** by the retrospective model, and represents the dam passage mortality for all dams and all years represented by *n*. μ_t is also **estimated** by the retrospective model and it represents incremental total mortality between the Snake River basin and the furthest up-river X-dam in year *t*.

The ultimate result of the retrospective analysis is a posterior probability distribution of estimates of both $m_{t,i}$ and $\Delta m_{t,i}$.

Prospective Implementation: In the prospective Delta model, the ($\Delta m_{t,i} - M_{t,i}$) term is combined and re-defined to accommodate three "extra mortality" hypotheses. Four estimates from the CRiSP and FLUSH combined passage and transportation models are **input** to the prospective model to allow estimation of this term:

$V_{n,t,i}$ = Direct Lower Granite pool to Bonneville Dam tailrace in-river survival (*n* refers to non-transported smolts) in year *t*.

$M_{t,i}$ = As defined above: direct survival of combined transported and non-transported smolts to below Bonneville Dam.

$P_{t,i}$ = The proportion of smolts survivng to below Bonneville Dam that were transported.

$D_{t,i}$ = The ratio of post-Bonneville survival of transported to non-transported smolts.

Prospective Implementation of the $(\Delta m_{t,,i} - M_{t,i})$ *Term For the "Hydro" Extra Mortality Hypothesis:*

In prospective analyses, the passage model terms identified above are identical for all Snake River sub-basins *i*, so this subscript is deleted from further descriptions for convenience. The representation is:

$$(\Delta m_{t,} - M_t) = -m_r + \ln(\omega_y / \omega_r) + \ln(\lambda_{n,y} / \lambda_{n,r})$$

in which the subscript *y* represents a prospective year (chosen from 1977-1992 water years, weighted to reflect 50-year water record), *r* represents a retrospective year (1977-1992) that matches the prospective water year, *n* represents non-transported fish, and

$\omega_r = \exp[-M_r]\,[D_r P_r + 1 - P_r]$

$\omega_y = \exp[-M_y]\,[D_y P_y + 1 - P_y]$

$\lambda_{n,r} = \exp[-m_r - \ln(\mathrm{T}_r)]$ and

$\lambda_{n,y} = 1-[(1-8_{n,r}) * ((1-V_{n,y})/(1-V_{n,r}))]$.

Prospective Implementation of the $(\Delta m_{t,,i} - M_{t,i})$ *Term For the "BKD" Extra Mortality Hypothesis:*

For the "BKD" extra mortality hypothesis, it is assumed that

$$\lambda_{n,y} = \lambda_{n,r}$$

so the representation is

$$(\Delta m_{t,} - M_t) = -m_r + \ln(\omega_y / \omega_r)$$

with all terms defined as in the "Hydro" extra mortality hypothesis representation.

Prospective Implementation of the $(\Delta m_{t,,i} - M_{t,i})$ *Term For the "Regime Shift" Extra Mortality Hypothesis:*

The representation is:

$$(\Delta m_{t,} - M_t) = -m_r + \ln(\omega_y / \omega_r) + \ln(\lambda_{n,y} / \lambda_{n,r})$$

in which terms are identical to the "Hydro" extra mortality implementation, with the exception of the subscripts *y* and *r* for estimation of the $\lambda_{n,y}$ term. For this term, the prospective water year *y* is matched with a retrospective year *r* that is in the same phase of an assumed 60-year climate cycle. For example, until brood year 2005 (relatively poor climate), the coupled brood years are chosen from retrospective brood years 1975-1990, then from prospective brood year 2006 for the next 30 years, the coupled retrospective years are chosen from brood years 1952-1974 (relatively good climate).

δ_t = common Snake River and lower Columbia River stock year–effect parameter for year *t*.

Retrospective Implementation: This parameter is **<u>estimated</u>** by the retrospective modeling procedure. The result is a posterior probability distribution of estimates.

Prospective Implementation: This parameter is **<u>input</u>** to the prospective model. Estimates are drawn from the posterior probability distribution generated by the retrospective model. The method by which they are selected depends upon the hypothesis regarding future climate that is under consideration.

Prospective Implementation of the δ Term For the Markov (Autoregressive) Future Climate Hypothesis

Because common year-effect estimates by the Delta model are similar in adjacent years (i.e., good years tend to follow good years and bad years tend to follow bad years), a Markov process with empirical probability densities to capture this autocorrelation was implemented. Details of the method are described in Deriso (1997) and Marmorek et al. (1998a, p. A116-A117).

Prospective Implementation of the δ Term For the Cyclical Future Climate Hypothesis

This approach assumes that common year-effect estimates of the Delta model follow a cyclical pattern suggested by inter-decadal climate shifts. This is modeled as a sine-wave crossing zero in brood year 1980, with an 18.5-year period. This is applied as a Markov process with details described in Deriso (1997) and Marmorek et al. (1998a, p. A117-A118).

$\varepsilon_{t,i}$ = normally distributed mixed process error and recruitment measurement, which depends on year t and sub-basin i.

Retrospective Implementation: This parameter is **<u>estimated</u>** by the retrospective modeling procedure. The result is a posterior probability distribution of estimates.

Prospective Implementation: This parameter is **<u>input</u>** to the prospective model. Estimates are drawn from the posterior probability distribution generated by the retrospective model. In prospective implementation, the process error variance is deflated to 61% of the posterior variance contained in the retrospective modeling results to account for confounding by observation error. Details are described in Deriso (1997).

Alpha Model Description

The Alpha model is described in Anderson and Hinrichsen (1997), Marmorek et al. (1998a, p. A91-92), Marmorek et al. (1998c, p. 54-55), and Hinrichsen and Paulsen (1998). The basic equation for the Alpha model is:

$$\ln(R_{t,i}) = (1+p)\ln(S_{t,i}) + a_i - b_i S_{t,i} - M_{t,i} - \alpha_{t,j} + \varepsilon_{t,i}$$

All terms in the Alpha model except the prospective implementation of M_t and prospective and retrospective implementation of $\alpha_{t,j}$ are identical to terms in the Delta model. Note that, while the Ricker a_i term is defined and estimated in a similar manner, it is not directly comparable to the Ricker a_i term estimated by the delta model because of the subtraction of averages in the $\alpha_{t,j}$ term (see below). Adjustment of the alpha model Ricker a_i term by addition of averages in the $\alpha_{t,j}$ term is necessary to make the alpha and delta model Ricker a_i terms comparable.

$M_{t,i}$ = direct passage mortality, which depends on year and includes combined mortality of both transported and non-transported smolts. For all sub-basins *i* within the Snake River sub-region, mortality is from the head of Lower Granite pool to below Bonneville Dam.

Prospective Implementation: This survival rate is <u>**input**</u> to the prospective model from FLUSH and CRiSP passage model estimates.

$\alpha_{t,j}$ = extra mortality in year *t* for subregion *j*. PATH analyses referred to in this appendix apply only to the Snake River subregion, although some PATH analyses have also estimated separate α's for the lower Columbia River subregion..

Retrospective Implementation:

$$\alpha_{i,j} = \alpha_n - [\text{average } \alpha_n] - \ln(D_t P_t + 1 - P_t) + [\text{average } \ln(D_t P_t + 1 - P_t)]$$

in which the averaged terms encompass brood years 1952-1990 and

$$\alpha_n = (c_1 / F_t) + (c_2 E_t / F_t) + STEP_j$$

This term is <u>**estimated**</u> in the retrospective model from other terms in the model and from the following additional values, which are <u>**input**</u> to the retrospective model:

$P_{t,i}$ = The proportion of smolts surviving to below Bonneville Dam that were transported.

$D_{t,i}$ = The ratio of post-Bonneville survival of transported to non-transported smolts.

F_t = Average flow (in kcfs) at Astoria for year t during April and June

E_t = Climate index variable (PAPA drift). This represents the latitude of a drifting object after three months drift starting at station PAPA.

$STEP_i$ for years prior to 1975 = zero. This term represents a 1975 brood year climate regime shift, which has different effects in different regions.

The specific terms that are **estimated** in the model are:

c_1, c_2 = estimated coefficients

$STEP_i$ for years subsequent to 1974 = estimated effect of climate regime shift occurring in 1975 brood year.

Prospective Implementation

In the prospective Alpha model, the $\alpha_{t,i}$ term is estimated in a manner consistent with each of three "extra mortality" and two "future climate" hypotheses. In addition to inputs described for the retrospective Alpha model, an additional **input** from the CRiSP and FLUSH passage models is:

$V_{n,t,i}$ = Direct Lower Granite pool to Bonneville Dam tailrace in-river survival (n refers to non-transported smolts) in year t.

Prospective Implementation For the "Hydro" Extra Mortality Hypothesis:

This implementation is identical to that in the prospective Alpha model, except for the value of *STEP* in any prospective year y:

$$STEP_y = -\ln[1-(1-\exp(-STEP_r))(1-V_{n,y}) / (1 - \text{average } V_{n,r})]$$

The average $V_{n,r}$ is estimated from 1975-1990 brood years and each retrospective year r represents a water year identical to that in each prospective year y. The prospective F, E variables are defined according to the particular climate hypothesis (see below).

Prospective Implementation For the "BKD" Extra Mortality Hypothesis:

In this implementation, $STEP_y = STEP_r$, therefore the equation is identical to the retrospective equation with $t = y$. The prospective F, E variables are defined according to the particular climate hypothesis (see below).

Prospective Implementation For the "Regime Shift" Extra Mortality Hypothesis:

For the regime shift extra mortality hypothesis, the $STEP_y$ value chosen for a given prospective year is one which occurred from the same phase of the cycle retrospectively. For example, until brood year 2005, $STEP_y$ is one drawn from brood years 1975-1990 (i.e., $STEP_y$). Then from 2006 for the next 30 years, $STEP_y$ = 0, which is the value applicable to retrospective brood years 1952-1974.

Prospective Implementation For the Markov (Autoregressive) Future Climate Hypothesis

A Markov process with empirical probability densities to capture adjacent year autocorrelations was implemented for the E_t PAPA index parameter. The value for F_y (Astoria flow in future year *y*) was chosen according to its negative correlation with unregulated water transit time (independent of future climate hypothesis). Details of the method are described in Deriso (1997) and Marmorek et al. (1998a, p. A116-A117) and Marmorek et al. (1998c, p. 65).

Prospective Implementation For the Cyclical Future Climate Hypothesis

This approach assumes that the E_t PAPA index parameter of the Alpha model follows a cyclical pattern suggested by inter-decadal climate shifts. This is modeled as a sine-wave crossing zero in brood year 1975, with an 18.5-year period. This is applied as a Markov process with details described in Deriso (1997) and Marmorek et al. (1998a, p. A117-A118). The value for F_y (Astoria flow in future year *y*) was chosen according to its negative correlation with unregulated water transit time (independent of future climate hypothesis).

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