US Army Corps
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Walla Walla District

FINAL

Lower Snake River Juvenile Salmon Migration Feasibility Report/ Environmental Impact Statement

APPENDIX A

Anadromous Fish Modeling

February 2002

FEASIBILITY STUDY DOCUMENTATION

	Document Title
Lower Snake River Juvenile Salmon Migration Feasibility Report/Environmental Impact Statement	
Appendix A (bound with B)	Anadromous Fish Modeling
Appendix B (bound with A)	Resident Fish
Appendix C	Water Quality
Appendix D	Natural River Drawdown Engineering
Appendix E	Existing Systems and Major System Improvements Engineering
Appendix F (bound with G, H)	Hydrology/Hydraulics and Sedimentation
Appendix G (bound with F, H)	Hydroregulations
Appendix H (bound with F, G)	Fluvial Geomorphology
Appendix I	Economics
Appendix J	Plan Formulation
Appendix K	Real Estate
Appendix L (bound with M)	Lower Snake River Mitigation History and Status
Appendix M (bound with L)	Fish and Wildlife Coordination Act Report
Appendix N (bound with O, P)	Cultural Resources
Appendix O (bound with N, P)	Public Outreach Program
Appendix P (bound with N, O)	Air Quality
Appendix Q (bound with R, T)	Tribal Consultation and Coordination
Appendix R (bound with Q, T)	Historical Perspectives
Appendix S*	Snake River Maps
Appendix T (bound with R, Q)	Clean Water Act, Section 404(b)(1) Evaluation
Appendix U	Response to Public Comments

*Appendix S, Lower Snake River Maps, is bound separately (out of order) to accommodate a special 11 x 17 format.

The documents listed above, as well as supporting technical reports and other study information, are available on our website at http://www.nww.usace.army.mil/lsr. Copies of these documents are also available for public review at various city, county, and regional libraries.

STUDY OVERVIEW

Purpose and Need

Between 1991 and 1997, due to declines in abundance, the National Marine Fisheries Service (NMFS) made the following listings of Snake River salmon or steelhead under the Endangered Species Act (ESA) as amended:

- sockeye salmon (listed as endangered in 1991)
- spring/summer chinook salmon (listed as threatened in 1992)
- fall chinook salmon (listed as threatened in 1992)
- steelhead (listed as threatened in 1997).

In 1995, NMFS issued a Biological Opinion on operations of the Federal Columbia River Power System (FCRPS). Additional opinions were issued in 1998 and 2000. The Biological Opinions established measures to halt and reverse the declines of ESA-listed species. This created the need to evaluate the feasibility, design, and engineering work for these measures.

The Corps implemented a study (after NMFS' Biological Opinion in 1995) of alternatives associated with lower Snake River dams and reservoirs. This study was named the Lower Snake River Juvenile Salmon Migration Feasibility Study (Feasibility Study). The specific purpose and need of the Feasibility Study is to evaluate and screen structural alternatives that may increase survival of juvenile anadromous fish through the Lower Snake River Project (which includes the four lowermost dams operated by the Corps on the Snake River—Ice Harbor, Lower Monumental, Little Goose, and Lower Granite Dams) and assist in their recovery.

Development of Alternatives

The Corps' response to the 1995 Biological Opinion and, ultimately, this Feasibility Study, evolved from a System Configuration Study (SCS) initiated in 1991. The SCS was undertaken to evaluate the technical, environmental, and economic effects of potential modifications to the configuration of Federal dams and reservoirs on the Snake and Columbia Rivers to improve survival rates for anadromous salmonids.

The SCS was conducted in two phases. Phase I was completed in June 1995. This phase was a reconnaissance-level assessment of multiple concepts including drawdown, upstream collection, additional reservoir storage, migratory canal, and other alternatives for improving conditions for anadromous salmonid migration.

The Corps completed a Phase II interim report on the Feasibility Study in December 1996. The report evaluated the feasibility of drawdown to natural river levels, spillway crest, and other improvements to existing fish passage facilities.

Based in part on a screening of actions conducted for the Phase I report and the Phase II interim report, the study now focuses on four courses of action:

- Existing Conditions
- Maximum Transport of Juvenile Salmon

- Major System Improvements
- Dam Breaching.

The results of these evaluations are presented in the combined Feasibility Report (FR) and Environmental Impact Statement (EIS). The FR/EIS provides the support for recommendations that will be made regarding decisions on future actions on the Lower Snake River Project for passage of juvenile salmonids. This appendix is a part of the FR/EIS.

Geographic Scope

The geographic area covered by the FR/EIS generally encompasses the 140-mile long lower Snake River reach between Lewiston, Idaho and the Tri-Cities in Washington. The study area does slightly vary by resource area in the FR/EIS because the affected resources have widely varying spatial characteristics throughout the lower Snake River system. For example, socioeconomic effects of a permanent drawdown could be felt throughout the whole Columbia River Basin region with the most effects taking place in the counties of southwest Washington. In contrast, effects on vegetation along the reservoirs would be confined to much smaller areas.

Identification of Alternatives

Since 1995, numerous alternatives have been identified and evaluated. Over time, the alternatives have been assigned numbers and letters that serve as unique identifiers. However, different study groups have sometimes used slightly different numbering or lettering schemes and this has led to some confusion when viewing all the work products prepared during this long period. The primary alternatives that are carried forward in the FR/EIS currently involve the following four major courses of action:

Alternative Name	PATH[1/] Number	Corps Number	FR/EIS Number
Existing Conditions	A-1	A-1	1
Maximum Transport of Juvenile Salmon	A-2	A-2a	2
Major System Improvements	A-2’	A-2d	3
Dam Breaching	A-3	A-3a	4

[1/] Plan for Analyzing and Testing Hypotheses

Summary of Alternatives

The **Existing Conditions Alternative** consists of continuing the fish passage facilities and project operations that were in place or under development at the time this Feasibility Study was initiated. The existing programs and plans underway would continue unless modified through future actions. Project operations include fish hatcheries and Habitat Management Units (HMUs) under the Lower Snake River Fish and Wildlife Compensation Plan (Comp Plan), recreation facilities, power generation, navigation, and irrigation. Adult and juvenile fish passage facilities would continue to operate.

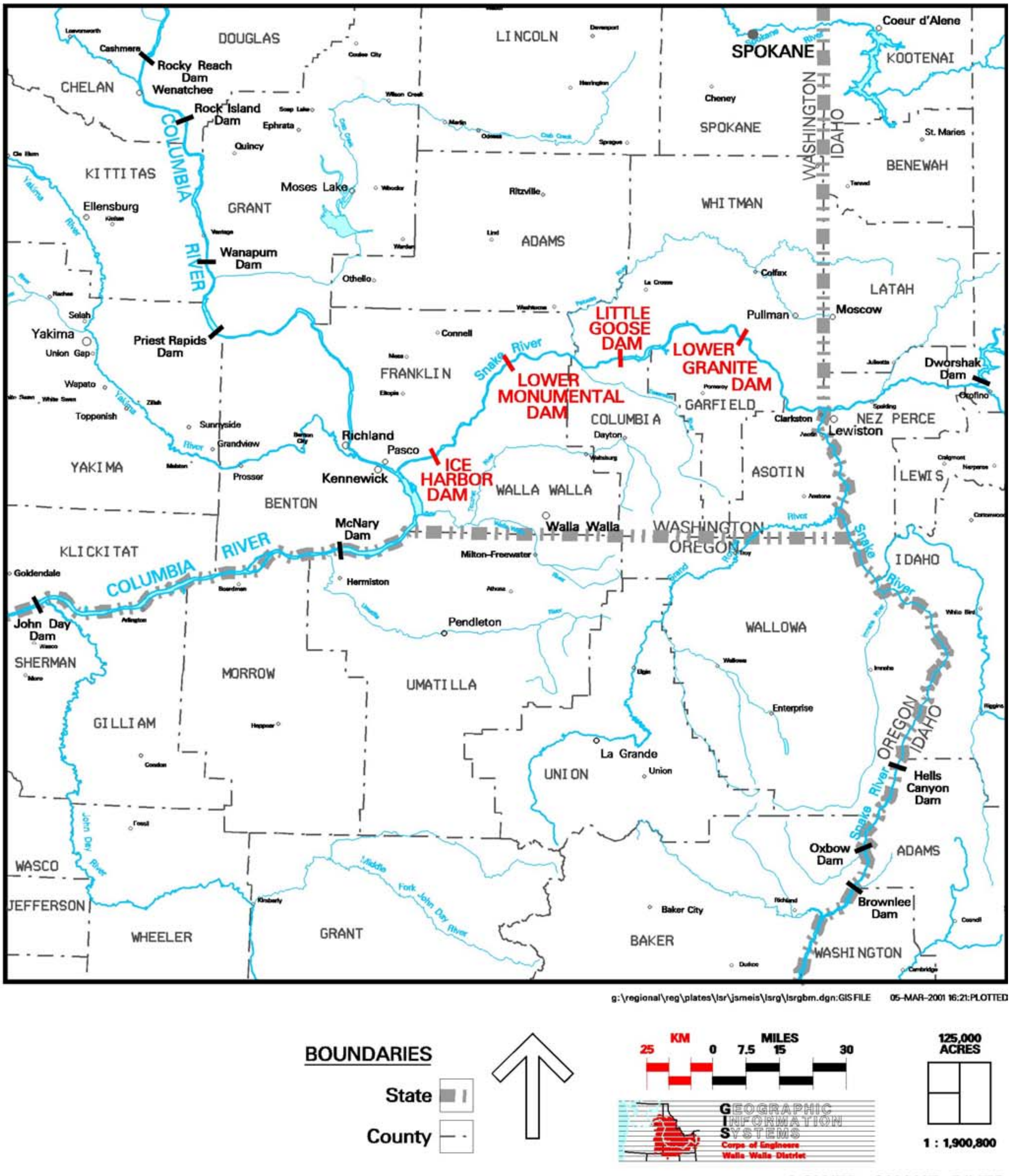
The **Maximum Transport of Juvenile Salmon Alternative** would include all of the existing or planned structural and operational configurations from the Existing Conditions Alternative. However, this alternative assumes that the juvenile fishway systems would be operated to maximize fish transport from Lower Granite, Little Goose, and Lower Monumental and that voluntary spill would not be used to bypass fish through the spillways (except at Ice Harbor). To accommodate this maximization of transport, some measures would be taken to upgrade and improve fish handling facilities.

The **Major System Improvements Alternative** would provide additional improvements to what is considered under the Existing Conditions Alternative. These improvements would be focused on using surface bypass facilities such as surface bypass collectors (SBCs) and removable spillway weirs (RSWs) in conjunction with extended submerged bar screens (ESBSs) and a behavioral guidance structure (BGS). The intent of these facilities would be to provide more effective diversion of juvenile fish away from the turbines. Under this alternative, an adaptive migration strategy would allow flexibility for either in-river migration or collection and transport of juvenile fish downstream in barges and trucks.

The **Dam Breaching Alternative** has been referred to as the "Drawdown Alternative" in many of the study groups since late 1996 and the resulting FR/EIS reports. These two terms essentially refer to the same set of actions. Because the term drawdown can refer to many types of drawdown, the term dam breaching was created to describe the action behind the alternative. The Dam Breaching Alternative would involve significant structural modifications at the four lower Snake River dams, allowing the reservoirs to be drained and resulting in a free-flowing yet controlled river. Dam breaching would involve removing the earthen embankment sections of the four dams and then developing a channel around the powerhouses, spillways, and navigation locks. With dam breaching, the navigation locks would no longer be operational and navigation for large commercial vessels would be eliminated. Some recreation facilities would close while others would be modified and new facilities could be built in the future. The operation and maintenance of fish hatcheries and HMUs would also change, although the extent of change would probably be small and is not known at this time.

Authority

The four Corps dams of the lower Snake River were constructed and are operated and maintained under laws that may be grouped into three categories: 1) laws initially authorizing construction of the project, 2) laws specific to the project passed subsequent to construction, and 3) laws that generally apply to all Corps reservoirs.



LOWER SNAKE RIVER
Juvenile Salmon Migration Feasibility Study

REGIONAL BASE MAP

US Army Corps of Engineers®
Walla Walla District

Final
Lower Snake River Juvenile Salmon Migration Feasibility Report/ Environmental Impact Statement

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Appendix A
Anadromous Fish Modeling

Produced by
National Marine Fisheries Service

Produced for
U.S. Army Corps of Engineers
Walla Walla District

February 2002

FOREWORD

Appendix A is the National Marine Fisheries Service Options Report "An Assessment of Lower Snake River Hydrosystem Alternatives on Survival and Recovery of Snake River Salmonids" dated October 1999. The NMFS report has been reformatted for consistency with other appendices. This appendix is one part of the overall effort of the U.S. Army Corps of Engineers (Corps) to prepare the Lower Snake River Juvenile Salmon Migration Feasibility Report/Environmental Impact Statement (FR/EIS).

The Corps has reached out to regional stakeholders (Federal agencies, tribes, states, local governmental entities, organizations, and individuals) during the development of the FR/EIS and appendices. This effort resulted in many of these regional stakeholders providing input and comments, and even drafting work products or portions of these documents. This regional input provided the Corps with an insight and perspective not found in previous processes. A great deal of this information was subsequently included in the FR/EIS and appendices; therefore, not all of the opinions and/or findings herein may reflect the official policy or position of the Corps.

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ACRONYMS AND ABBREVIATIONS

AIC	Aikiaki Information Criteria
BKD	bacterial kidney disease
BOR	U.S. Bureau of Reclamation
BPA	Bonneville Power Administration
BRD	Biological Resources Division
BRWG	Biological Requirements Work Group
BRZ	boat restricted zone
BY	Broad Year
CBFWA	Columbia Basin Fish and Wildlife Authority
Corps	U.S. Army Corps of Engineers
CRI	Cumulative Risk Initiative
CRiSP	Columbia River Salmon Passage
CWT	coded wire tag
ELBS	extended-length submerged bar screen
ESA	Endangered Species Act
ESU	evolutionarily significant unit
FCRPS	Federal Columbia River Power System
Feasibility Study	Lower Snake River Juvenile Salmon Migration Feasibility Study
FGE	fish guidance efficiency
FLUSH	Fish Leaving Under Several Hypotheses
FWCAR	Fish and Wildlife Coordination Act Report
HYURB	Hanford/Yakima Upriver Brights
IDEQ	Idaho Department of Environment Quality
IDFG	Idaho Department of Fish and Game
ISG	Independent Scientific Group
kcfs	thousand cubic feet per second
LSTS	lowered submerged traveling screen
NMFS	National Marine Fisheries Service
NRC	National Research Council
NWFSC	Northwest Fisheries Science Center
ODFW	Oregon Department of Fish and Wildlife
PATH	Plan for Analyzing and Testing Hypotheses
PDO	Pacific Decadal Oscillation
PIT	passive integrated transponder
PSC	Pacific Salmon Commission
SAR	smolt-to-adult return
SBTOC	Stanley Basin Technical Oversight Committee
s_e	survival below Bonneville Dam
SRB	Snake River bright
SRP	Scientific Review Panel
SRT	Scientific Review Team
STEP	Salmon and Trout Enhancement Program

ACRONYMS AND ABBREVIATIONS

STS	submerged traveling screen
TAC	technical advisory committee
TCR	transport:control ratio
UI	University of Idaho
URB	upriver brights
USFS	U.S. Forest Service
USFWS	U.S. Fish and Wildlife Service
WDFW	Washington Department of Fish and Wildlife

Executive Summary

ES.1 Overview of Analytical Approaches

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 appendix represents 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 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 downriver passage, and growth and survival in the ocean to 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 used in these analyses are time series of fish numbers in different life stages, as well as more focused experimental studies using marked and passive integrated transponder (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) used two complementary analytical approaches: (1) the Plan for Analyzing and Testing Hypotheses (PATH) analysis and (2) the Cumulative Risk Initiative (CRI) analysis.

ES.1.1 The PATH Analysis

PATH refers to a multi-agency, multi-participant process. The mechanics of the analysis embedded in the PATH process are technically difficult, but simple to understand in principle. Specifically, PATH applies a life-cycle model to historical data and:

- establishes estimates of historical trends in reproduction and components of survival (such as inriver 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 the migration corridor
- generates estimates of variability in the underlying processes.

In the second stage of analysis, PATH uses the 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 interpreted in light of sensitivity analyses (different visions of how factors outside the hydropower system might change in the future). The uncertainty in model output comes from the inherent uncertainty of a variable environment (e.g., the next few years could bring drought or high rainfall) and from different assumptions invoked when running the model. Although PATH examines as many as six or seven different management options, this report focuses primarily on comparisons of the breaching of four dams on the lower Snake River versus no breaching (but transportation of fish in barges). This Executive Summary refers to these options simply 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. Of the many PATH outputs, NMFS focuses on the frequency of computer simulation runs that meet particular survival and recovery criteria. The actual criteria depend on the characteristics of each stock and its natal stream, but can be thought of as low-population threshold requirements to be exceeded in more than 70 percent of the years (survival criterion), and an upper-population threshold to be 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. The average of these fractions over all assumption sets provides an average measure of the success of an action. An alternative way of summarizing the same data is to use the percentage of assumption sets in which survival and recovery criteria are met.

ES.1.2 The CRI Analysis

To complement PATH, NMFS has recently undertaken an additional analytical approach, CRI. In designing this complementary CRI approach, NMFS sought to address three factors not specifically examined in the PATH analyses:

1) CRI provides estimates of the risk of extinction faced by populations; PATH analyses do not estimate this risk.
2) CRI uses the annual rate of population growth as the performance measure for all management actions and translates projected changes in this annual rate into predicted changes in extinction risk.
3) CRI was designed to examine and compare possible benefits that might result from many different mixes of management actions; PATH analyses are more focused on hydrosystem improvements.

The CRI approach cannot replace the detailed examination PATH provides of modifications flow regimes, transport systems, or fish passage systems, and it is not intended to do so. Rather, CRI offers a concise assessment of broad arrays of management options by breaking the analyses into the following steps:

1) estimate the annual rate of population change under current conditions, and from that rate, calculate the risk of extinction for index stocks
2) construct demographic projection matrices that depict current demographic performance and calculate a predicted annual rate of population growth under current conditions
3) perform sensitivity analyses to assess where the greatest opportunities for promoting recovery exist in the life cycles of threatened salmonids (Note that an important next step will be to assess the biological feasibility of achieving improvements at identified life stages through specific actions.)

4) manipulate the baseline current matrices in ways that simulate hypothesized effects of management actions, and calculate the percent increase in annual population growth rate associated with each management experiment
5) relate increases in average population growth rates back to reductions in extinction risk
6) explore whether the connection between the management action and the hypothesized demographic response is biologically feasible, for those management experiments that are numerically effective
7) place all data used in analyses and examples of analyses on a public Web site so that others can repeat analyses or perform alternative analyses.

A major philosophical difference between CRI and PATH analyses is that the CRI analysis separates sensitivity analyses and numerical experiments from the question of what is biologically feasible. In contrast, the PATH analyses implicitly link numerical experiments and feasibility assessments into one large set of modeling runs.

ES.1.2.1 Changes to the CRI Analysis in the Final Anadromous Fish Appendix

Numerous changes to the CRI analyses have been made since the Draft Anadromous Fish (A-fish) Appendix was released in 1999.

For extinction risk analyses, including the determination of annual population growth rates, the following changes were made:

- All population counts were updated to include 1999 returns whenever possible.
- The risk of extinction is estimated in 24 and 100 years, for compatibility with PATH analyses and previous management decisions.
- CRI now uses a method of determining population growth rate (outlined in Chapter 8 of this appendix), which is robust to sampling error. This method uses a “running sum” of spawner counts, weighted by average ages of return, and provides an unbiased estimator of the average population growth rate.
- Because naturally spawning hatchery fish can potentially mask the true status of wild stocks, CRI analyses estimate annual population growth rates under several levels of hatchery fish reproductive success. In Chapter 8, the most extreme assumptions are presented (hatchery fish do not reproduce, and hatchery fish reproduce at a rate equal to wild fish). In addition, tables showing annual population growth rate, extinction risk, and needed improvements under two additional assumptions (hatchery fish reproductive success = 20 percent and 80 percent of wild fish) are also presented, for consistency with the 2000 FCRPS Biological Opinion.
- Annual population growth rates for several additional Snake River spring/summer chinook stocks were calculated. (It was not possible to calculate extinction risk for these stocks, because they are based only on redd counts, not total spawner populations.)
- The extinction risk threshold is now one fish in one generation (or true extinction). This threshold is possible due to the use of the running sum. This threshold was chosen because it is the most biologically meaningful and, therefore, comparable across stocks of different

inherent size or productivity. However, the extreme nature of this threshold should be borne in mind when evaluating risk.

- Evolutionarily significant unit-level population growth rates and risk of extinction were calculated using dam counts.

For matrix (life-cycle) analyses, several changes with respect to previous analyses were incorporated:

- The annual population growth rate (lambda, or dominant eigenvalue) was determined using recruit-per-spawner data from the most recent 5 years. This change was made to address the apparent increasing rate of decline over the most recent years.
- Estuarine and early ocean survival rates (s_e) were calculated from recent smolt-to-adult return rates (SARs). This resulted in lower estimates of s_e than previous analyses had used, and higher estimates of first-year survival (s_1). First-year survival rates fall within the range of published freshwater survival rates.
- As in previous analyses, CRI analyses evaluate the impact of indirect mortality attributable to the hydrosystem over a wide range of values.

ES.2 Key Uncertainties

One of the most fundamental uncertainties concerns the estimation of population trends for wild fish populations. While population trends superficially seem to be well-known, the presence of hatchery fish on natural spawning grounds introduces potentially enormous uncertainty (in proportion to the number of hatchery fish that show up spawning in the wild). The uncertainty arises because the offspring of hatchery fish get counted as recruits produced by wild fish, thereby potentially inflating estimates of the vitality of wild stocks. The uncertainty can be removed only if all hatchery fish are marked, and if sound estimates are available of the relative fitness of hatchery fish when spawning in the wild.

Assuming that the status and trends of wild salmon populations are known, the next key questions concern the causes of their endangerment. The decline of salmonid populations in the Snake River and elsewhere in the Pacific Northwest have coincided with a broad range of extensive environmental changes, including the construction of numerous dams, massive degradation of habitat quality, intense harvest, 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 salmonids, it clearly is not the only threat. There is a natural tendency to attribute the entire salmonid problem to dams, since dams are so massive and visible. The situation is not, however, that unambiguous, and considerable investments in transportation and bypass systems have clearly mitigated some of the harmful effects of dams with respect to salmonid mortality in the migration corridor. Because it is not possible to go back in history and to conduct experiments, it is impossible to definitively conclude where management should turn for salmon recovery. Therefore, a major uncertainty exists in the degree to which aspects of the ecosystem other than hydropower have contributed to declines in salmonid populations.

The biggest puzzle regarding the potential impacts of dams is this: after summing up the many fish that are transported downstream in barges and the fish that travel downstream through bypass and spill facilities, the direct mortality observed within the Snake River migration corridor is not sufficiently high to account for the poor smolt-to-adult returns of these stocks. This accounting dilemma has directed attention to the concept of "extra mortality." Extra mortality is the unexplained mortality of Snake River salmonids outside the migration corridor. A number of hypotheses for the cause of this extra mortality have been proposed: the hydropower system itself may weaken fish or disrupt their natural rhythms, leading to poor smolt-to-adult returns; hatcheries may interfere with the fitness and survival of wild fish; habitat degradation may reduce stock vigor; genetic effects may reduce stock viability; or ocean conditions may differentially affect salmonids that spawn above the Snake River dams. Similarly, although fish suffer almost no mortality during the process of being transported, once they are released below Bonneville Dam it is possible that these transported fish suffer their own special form of extra mortality, called "differential delayed transportation mortality." The problem with extra mortality and differential delayed transportation mortality is that it is not easy to quantify their magnitudes or to identify the causes of the mortality. Nonetheless, the value of alternative management actions hinges on estimates of differential delayed transportation mortality and on the hypothesized causes of extra mortality. For example, if differential delayed transportation mortality were large, then the removal of dams (which would eliminate the need for transportation of smolts) would result in greatly improved survival rates. Conversely, if differential delayed transportation mortality were low and extra mortality were not due to the hydropower system, then removal of dams would not significantly benefit Snake River stocks.

ES.3 Results from the PATH Analyses

ES.3.1 Spring/Summer Chinook Salmon

PATH results indicate that breaching, under a wide variety of assumptions, is more likely than transportation options to meet recovery and survival criteria for spring/summer chinook salmon. Specifically, the average fraction of simulation outputs that meet recovery and survival criteria is much larger if dams are breached than if transportation is the primary management tool (82 percent versus 47 to 50 percent). However, it is worth noting that all PATH prospective simulations indicate that all index stocks exhibit increasing population trends, even under the assumption of existing status-quo improvements to the hydropower system. Because run-reconstruction data reveal declining populations, the increasing trends predicted by PATH models (with or without dam breaching) suggest that PATH simulations may be too optimistic.

The PATH analyses clearly highlight the key uncertainty underlying the relative benefits to be accrued by dam breaching. Specifically, if one assumes differential delayed mortality is relatively low and indirect mortality of transported and inriver fish is unrelated to the hydrosystem experience, the advantage of breaching dams relative to transportation is greatly reduced.

ES.3.2 Fall Chinook Salmon

PATH results for fall chinook salmon parallel those for spring/summer chinook salmon. Across a wide range of assumptions and uncertainties, breaching is more likely to meet recovery criteria than transportation options. Again, however, the relative advantage of breaching compared to transportation is sensitive to assumptions about the magnitude of differential delayed transportation mortality. Dam breaching is predicted to increase escapement levels between 33 percent (no differential delayed transportation mortality) and over 1,000 percent (high differential delayed transportation mortality). Importantly, there is an additional route by which breaching is expected to benefit fall chinook salmon, without regard to any assumptions about differential delayed transportation mortality. Because fall chinook salmon spawn in the mainstem river as opposed to tributaries and streams, breaching is expected to increase the carrying capacity (available habitat) for fall chinook salmon by more than 70 percent.

ES.3.3 Steelhead

For Snake River steelhead, there are insufficient data to produce quantitative analyses at the level of detail possible for chinook salmon. Given the scarcity of data, one reasonable approach is to explore the extent to which steelhead behave like chinook salmon, and then use results from chinook salmon to draw conclusions about steelhead. This has to be done with caution, however, because while aspects of steelhead population trends mirror those of spring/summer chinook salmon, there are also notable discrepancies. In general, management actions that improve conditions for spring/summer chinook salmon are likely to also improve conditions for steelhead. However, management that does not lead to chinook salmon recovery might still recover steelhead stocks.

ES.3.4 Sockeye Salmon

There is an even greater absence of data for sockeye salmon in the Snake River than for steelhead. Currently, a captive broodstock program maintains the sockeye salmon populations in the Snake River. Numbers of natural spawners are so low that there are not prospects for generating life-cycle data of the caliber needed for a formal risk analysis or recovery and survival analysis.

ES.4 Results from the CRI Analyses

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 focused on actions and impacts related to the hydropower system. It is important to also ask, however, how broader changes and combinations of changes in salmonid management might affect the species-by-species conclusions enumerated above, as well as the conclusions about the relative merits of hydrosystem management actions. The CRI analysis is designed to consider a more comprehensive set of potential management actions. Clearly, if actions outside the hydrosystem could dramatically improve survival rates or productivity for listed species, then these suites of actions must be considered when evaluating dam breaching.

ES.4.1 Extinction Risks If Current Conditions Persist

The CRI analyzed population trends as revealed by spawner counts from 1980 until present. This time period was selected to represent a relatively stable period with respect to the construction and modification of the Snake River hydropower system. The method applied by CRI is robust to large and erratic sampling errors, which are likely to plague spawner census data. Using this approach, and assuming that hatchery fish have 20 to 80 percent of the reproductive success of wild fish, annual rates of population change for spring/summer chinook salmon index stocks ranged from 0.88 to 1.04. Annual rates of population change for Snake River fall chinook ranged from 0.87 to 0.92; annual rates of decline for Snake River steelhead ranged from 0.74 to 0.85. (Values less than 1.00 represent a declining population; greater than 1.00 indicate that the population is increasing.) These annual rates of population change, coupled with high variation in spawner abundance from year to year, translate into substantial risks of extinction, and require marked improvements in survival if they are to be reversed. Reduction in harvest rates (which are already reduced) could reverse the population decline for Snake River fall chinook salmon, and could improve (but not reverse) the situation with Snake River steelhead. However, harvest is so low on Snake River spring/summer chinook salmon that further harvest reductions offer negligible benefits. Spring/summer chinook salmon require the largest improvements in annual population growth to reduce extinction risk substantially. (However, Snake River steelhead require the largest improvements to achieve an increasing population growth rate. Data to evaluate extinction risk for populations of steelhead are missing.) The key management question is where in the spring/summer chinook life cycle these benefits could be realized, and via what management actions.

ES.4.2 Matrix Analyses of Which Life History Stages to Target

The next step in the CRI analyses is to construct demographic projection matrices that depict current demographic performance. Sensitivity analyses are then used to assess where the greatest opportunities for promoting recovery exist in the life cycles of threatened salmonids. For spring/summer chinook salmon, improvements in first-year survival and in survival upon entering the estuary and ocean would have the greatest impact on annual rates of population growth. In contrast, <u>further</u> engineering improvements in existing bypass and transportation systems have little likelihood of substantially increasing annual population growth. This does not mean that existing fish passage improvements and flow regime regulations are not important; indeed, analyses indicate that if the hydropower system had not been altered to facilitate fish passage and transportation, spring/summer chinook salmon would have declined precipitously.

Overall, the major uncertainty for the CRI analyses is the "biological feasibility" of achieving sufficient demographic improvements as a result of particular management actions. Harvest reduction is one management action for which the feasibility of achieving a specific demographic effect is clear. However, the demographic consequences of virtually every other management action are uncertain. The major uncertainties with respect to biological feasibility identified by the CRI analyses echo the uncertainty about extra mortality and differential delayed transportation mortality emphasized by the PATH analyses. For example:

- The benefits of breaching the four Snake River dams depend on how much the survival of fish below Bonneville Dam is expected to increase after dams are breached.
- Whether maximum transportation could recover stocks depends on the extent to which transported fish suffer additional mortality below Bonneville Dam as a result of being transported.

However, the CRI analyses point to some additional uncertainties that warrant much more study than has been completed to date. In particular, studies are needed to:

- Quantify the connection between habitat quality and salmon productivity (since land use patterns, management of the hydropower system, and pollution all influence habitat quality)
- Assess the biological mechanisms underlying the linkages between ocean conditions and the survival and growth of adult salmon

- Pay greater attention to hatchery releases as a mortality factor that might be reduced through alterations in hatchery programs
- Investigate the possibility of increasing estuarine survival (a lifestage with major impacts on annual population growth rates) by reducing predators, such as Caspian terns, at the mouth of the Columbia River
- Explore the possibility that low smolt-to-adult returns are due to sub-lethal reductions in fitness, which, although not easily detected in survival studies, could be reversed by management actions.

NMFS has recently 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. It will require anywhere from 2 to 10 years for these studies to provide information about the feasibility of achieving demographic improvements through different management actions. Given the substantial short-term extinction risks, it may be useful to initiate some management actions or "experiments," even if the actions are not certain to reap substantial benefits.

ES.5 Conclusions

1) PATH analyses suggest that breaching is more likely than any other change in the hydropower system to meet survival and recovery criteria for the listed species across the widest range of assumptions and scenarios. However, the PATH analyses did not determine whether breaching is necessary and/or sufficient for recovery.

2) CRI matrix analyses indicate that improvements in inriver survival cannot by themselves reverse population declines in Snake River spring/summer chinook salmon. However, past improvements have greatly reduced rates of decline. Under current conditions, reductions in mortality on the order of 5 to 10 percent are needed in the estuarine environment, or in the first year of life. What this means for the question of dam breaching, is that if the removal of four Snake River dams is to reverse the population decline in Snake River spring/summer chinook salmon by itself, it will have to result in the survival of roughly 5 to 10 of every 98 smolts that are currently dying in the estuary.

3) CRI analyses conclude that further improvements in spill and bypass systems or in transportation are unlikely to be adequate to rebuild the threatened and endangered Snake River salmonid populations.

4) Both the PATH and CRI analyses highlight differential delayed transportation mortality and extra mortality as critical uncertainties in the analyses. The efficacy of dam breaching for spring/summer chinook salmon recovery is strongly affected by these factors.

5) The CRI analyses highlight an additional suite of critical uncertainties due to lack of data, including the possibility of attaining increased productivity with habitat management and of enhancing survival via improved hatchery practices or the control of salmonid predators.

6) The CRI analyses emphasize that apart from uncertainty about the effectiveness of different management actions, there is also uncertainty about the status and trend of wild salmon populations. The reason for this most basic uncertainty is uncertainty about the contribution hatchery fish make to recruits to natural spawning grounds.

1. Introduction

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 appendix focuses on an ecological assessment of management alternatives for the Federal Columbia River Power System (FCRPS).

The National Marine Fisheries Service (NMFS) 1995 FCRPS Biological Opinion (NMFS, 1995a) 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 facilities on the lower Snake River. The purpose of this evaluation is to determine the likelihood that drawdown (breaching) of these four facilities, 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 Salmon Migration Feasibility Study (Feasibility Study), the U.S. Army Corps of Engineers (Corps) requested that NMFS 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 that 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 hydropower system (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 it.

After a brief preview of the general salmonid life cycles and key issues surrounding salmonid recovery (Section 2), this document has two main analytical portions. The first portion represents the Plan for Analyzing and Testing Hypotheses (PATH) analytical framework (described in Section 3) applied to spring/summer chinook salmon (Section 4), fall chinook salmon (Section 5), steelhead (Section 6), and sockeye salmon (Section 7). The next major portion of the report applies a complementary analytical framework, the Cumulative Risk Initiative (CRI), to spring/summer chinook salmon, fall chinook salmon, and steelhead (Section 8). The next section contains updates on differential delayed transportation mortality and research related to dam passage (Section 9). The final section (Section 10) reconciles the different views offered by these alternative decision-making analytical tools and summarizes the key implications for management alternatives.

2. The Ecology, Ecological Risks, and Uncertainties Surrounding Salmon in the Snake River

2.1 Historical Trends

The Snake River historically was and currently 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 19th 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 began at the end of the 19th century as a result of overfishing; by early in the 20th century, however, environmental degradation from mining, grazing, logging, and agriculture caused further declines. Before 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 percent 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 that 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 (141 and 182 miles) of mainstem habitat, respectively. This changed the lower mainstem river from a mostly free-flowing body into a series of reservoirs covering about 70 percent 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 (National Research Council [NRC], 1996).

2.2 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).

2.2.1 Adult Stage

Salmon originating in the Snake River reside in the ocean from months to years, depending on 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.

2.2.2 Egg-to-Smolt Stage

Salmon eggs are deposited in excavated nests called redds and are covered with a layer of gravel. The eggs incubate in the gravel over winter, with the young salmon hatching and migrating 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 to 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.

2.2.3 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 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 facilities 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 is collected at the uppermost 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.

2.2.4 Estuarine/Early Ocean Stage

Like salmon runs from other parts of the Columbia River Basin, Snake River salmon depend 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 portion of the production from a particular brood year (jacks and minijacks) returns to the Columbia River after a few months to 1 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 2, 3, 4, or more years at sea, and the cycle continues.

2.3 Qualitative Overview of the Likely Effects of the Hydropower System on Anadromous Salmonids

In assessing the potential effects of alternative hydropower options on listed Snake River salmonids, NMFS has focused primarily on quantitative analyses. A complementary discussion of alternative management options can be found in the U.S. Fish and Wildlife Service Coordination Act Report (FWCAR, 1999). In lieu of repeating the FWCAR, this report briefly discusses the many hypothesized effects of hydropower operations on salmonids, with the intent of sketching the big picture as opposed to discussing all the details.

It is important to recognize that the hydropower system can have many potential impacts on salmonids. The most obvious impact is that dams obstruct fish passage during downstream migration and again later in their life cycle when the fish return to spawn. The direct effects of this obstruction have been well measured and are substantial. Offsetting these direct effects are improved dam bypass systems, fish ladders, and transportation of fish in barges. To examine ONLY directly measured mortality, it appears that transportation systems and bypass systems either offset or come close to offsetting the direct losses (largely because of high downriver survival for barged fish from the point of collection to the point of release below Bonneville Dam). It is critical to realize the hydropower system includes more than mainstem dams. Storage reservoirs have modified flows, and the quality of water (turbidity and sediment loads) may be altered in ways that modify conditions where the river meets the ocean.

The less obvious effects of the hydropower system include habitat loss (due to flooding created by reservoirs, which is an issue largely for fall chinook salmon), altered environments with respect to nutrient replenishment and predators, and potentially reduced fitness of fish. The hypothesis of reduced fitness of fish has proven elusive. It is certainly plausible that when a natural river system is drastically altered, as it is when several dams are placed on a river, then the organisms that evolved in the natural river system may suffer subtle reductions in their fitness that do not appear as mortality during actual mainstem migration. For instance, operation of the hydropower system has created changes that lead fish to experience different thermal regimes; this is significant because temperature is known to have an impact on salmonid fitness, without necessarily causing immediate mortality. This reduced fitness, or latent mortality expressed outside the hydropower corridor (but caused by the hydropower system), has not been directly quantified with any success. This does NOT mean it does not exist—only that data unambiguously and directly documenting its magnitude are lacking.

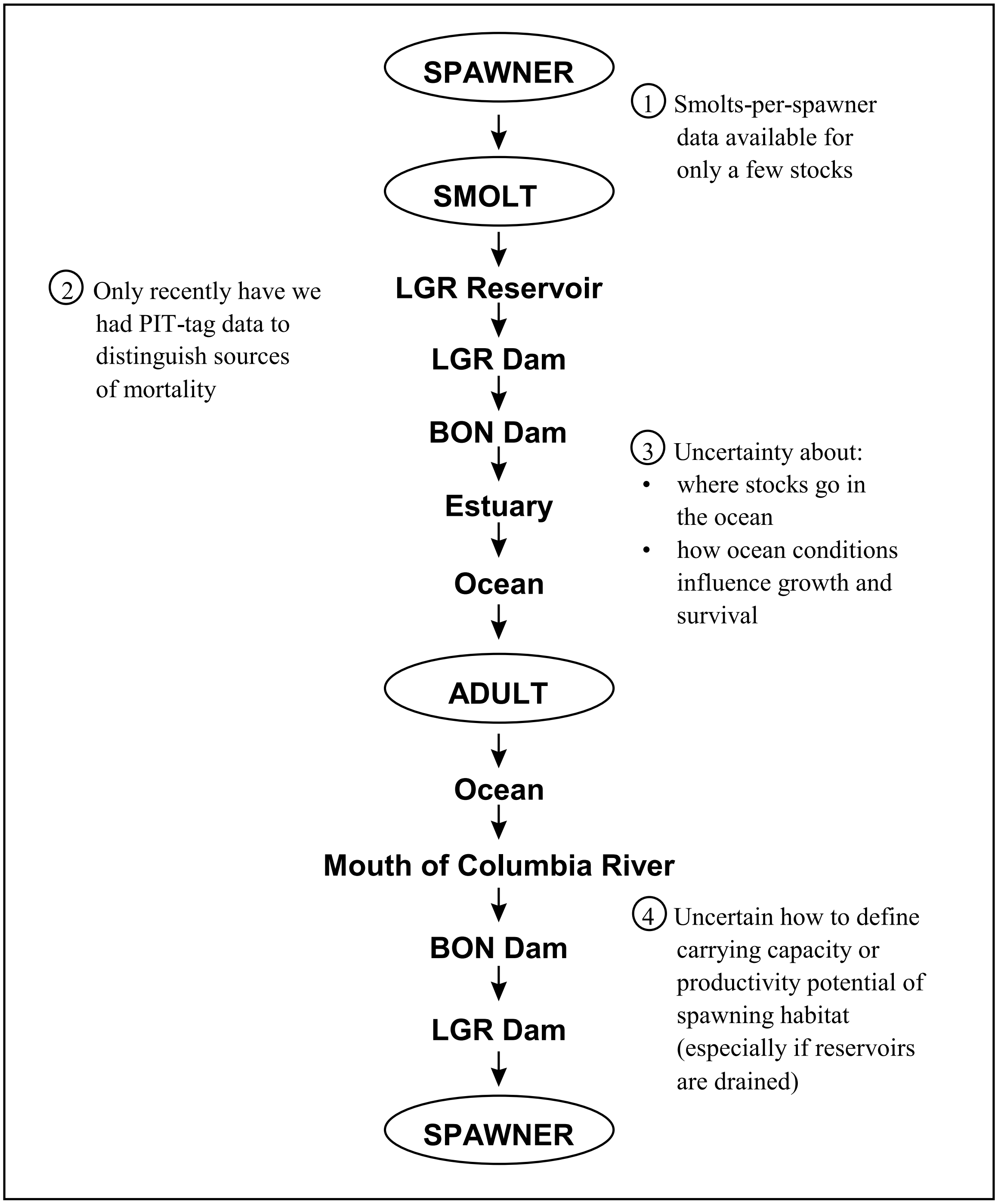
Attempts to estimate latent reduced fitness indirectly have been made in the context of models (as a factor explaining residual variation after accounting for direct mortality and density-dependent recruitment). The primary evidence supporting reduced fitness has been the more rapid decline of stocks above the four Snake River dams than of stocks below the Snake River dams, even after accounting for direct mortality to the hydropower system. But comparing upriver and downriver stocks is not clear cut; for example, the recruits-per-spawner ratios for upriver stocks did decline following construction of the Snake River dams, but the decline did not occur until 7 or 8 brood years after the dams were completed. The two analytical frameworks discussed in this report adopt different approaches to this uncertainty: the PATH framework (introduced in Section 3) attempts to estimate latent mortality due to the hydropower system by performing a series of population dynamics model-fitting exercises and offering different hypotheses that might explain residual variation (one set of hypotheses corresponding to reduced fitness caused by the hydropower system). The CRI framework, introduced in Section 8, leaves the question of this latent mortality open and simply simulates different future scenarios assuming different amounts of mortality below Bonneville Dam that might be relieved if dams were breached. The challenge raised by the CRI exercise is prompting scientists to obtain direct data regarding to potential magnitude of latent mortality due to the hydropower system.

In the following sections, some of the key technical intricacies and issues surrounding the quantitative assessment of the effects of hydropower system effects are discussed. Many of the issues have their own jargon as a result of words coined during the PATH process. NMFS uses this jargon but explains the terms in other words as well.

2.4 Previewing the Key Uncertainties

2.4.1 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 (before 1970), when salmonid populations were also much higher. Scientific complexity arises because many environmental factors 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, 1996b). One way of tackling this problem is to associate past changes with blame—in other words, identify particular components of the fish life cycle (see Figure 2-1) 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



① Smolts-per-spawner data available for only a few stocks

② Only recently have we had PIT-tag data to distinguish sources of mortality

③ Uncertainty about:

- where stocks go in the ocean
- how ocean conditions influence growth and survival

④ Uncertain how to define carrying capacity or productivity potential of spawning habitat (especially if reservoirs are drained)

Note: 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 make assumptions about quantitative changes in survival at these steps.

Figure 2-1. Straight-Line Representation of a Generalized Life Cycle of Snake River Salmonids in practice. First, to extend the analogy, the patient's symptoms are consistent with those of many different diseases. In other words, many factors potentially affect 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; and increased logging, harvest, acres of irrigated lands, and so forth (NRC, 1996b). Similar correlations exist on the finer scale of Snake River salmon stocks, which are well illustrated simply by displaying the population trajectories or trends in smolt-to-adult returns for spring/summer chinook salmon in conjunction with number of dams (Figure 2-2), total hatchery releases (Figure 2-3), or indices of ocean conditions (Figure 2-4). 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, 1996b).

Before discussing specific analyses, this section introduces key technical ideas that contribute to the scientific debate surrounding strategies for salmonid recovery and that provide a foundation for understanding particular analyses. To help the reader, a glossary of frequently used technical terms is provided in Table 2-1. Although not all of the terms in this glossary are discussed in this section of the report, this glossary is intended to be a convenient reference for terms used throughout the report.

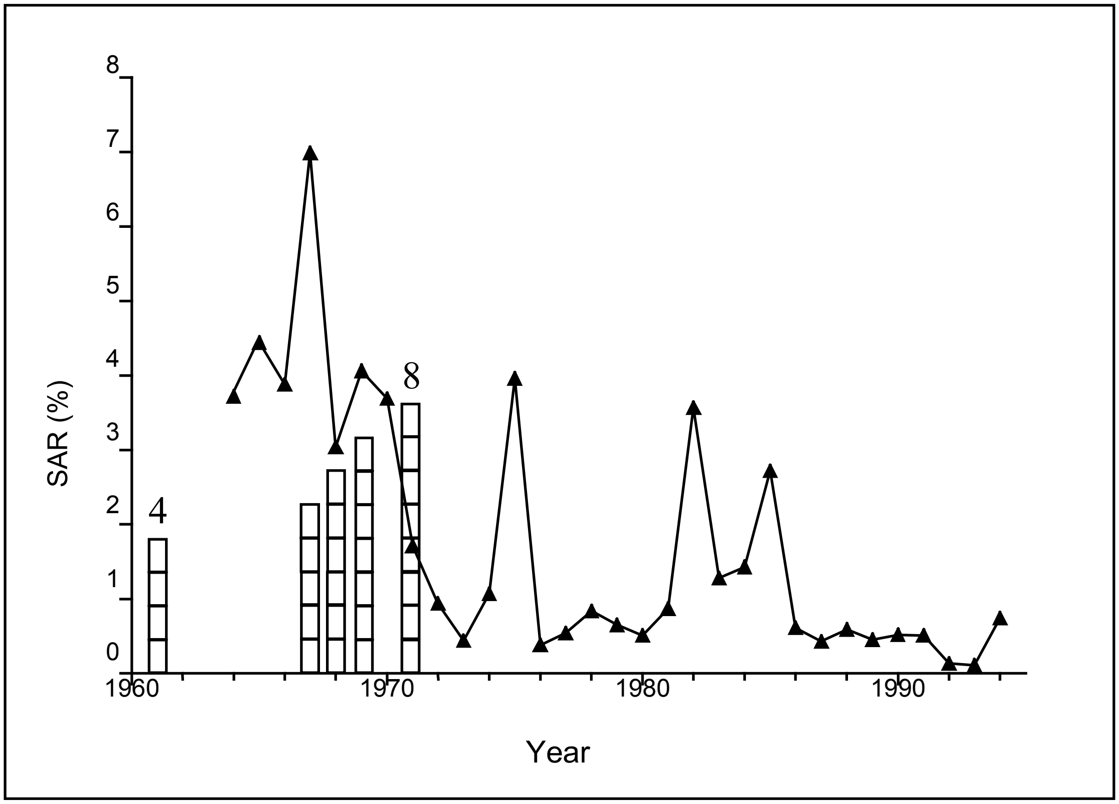
2.4.2 Differential Delayed Transportation Mortality

Many fish are transported to below the Bonneville Dam in barges (e.g., between 50 and 60 percent of the spring/summer chinook salmon in 1996 and 1997; Marmorek et al., 1998). 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 not 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 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 corrected readily by the removal of dams. Estimates of differential delayed transportation mortality have been made for outmigration years spanning two decades. Scientists differ on which estimates of differential delayed transportation mortality they believe should be given the greatest credence. The parameter of interest in this debate is the D-value (the ratio of survival below Bonneville Dam for transported fish compared to untransported fish); $D = 1$ 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 inriver migrants once all the fish are below Bonneville Dam).

2.4.3 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

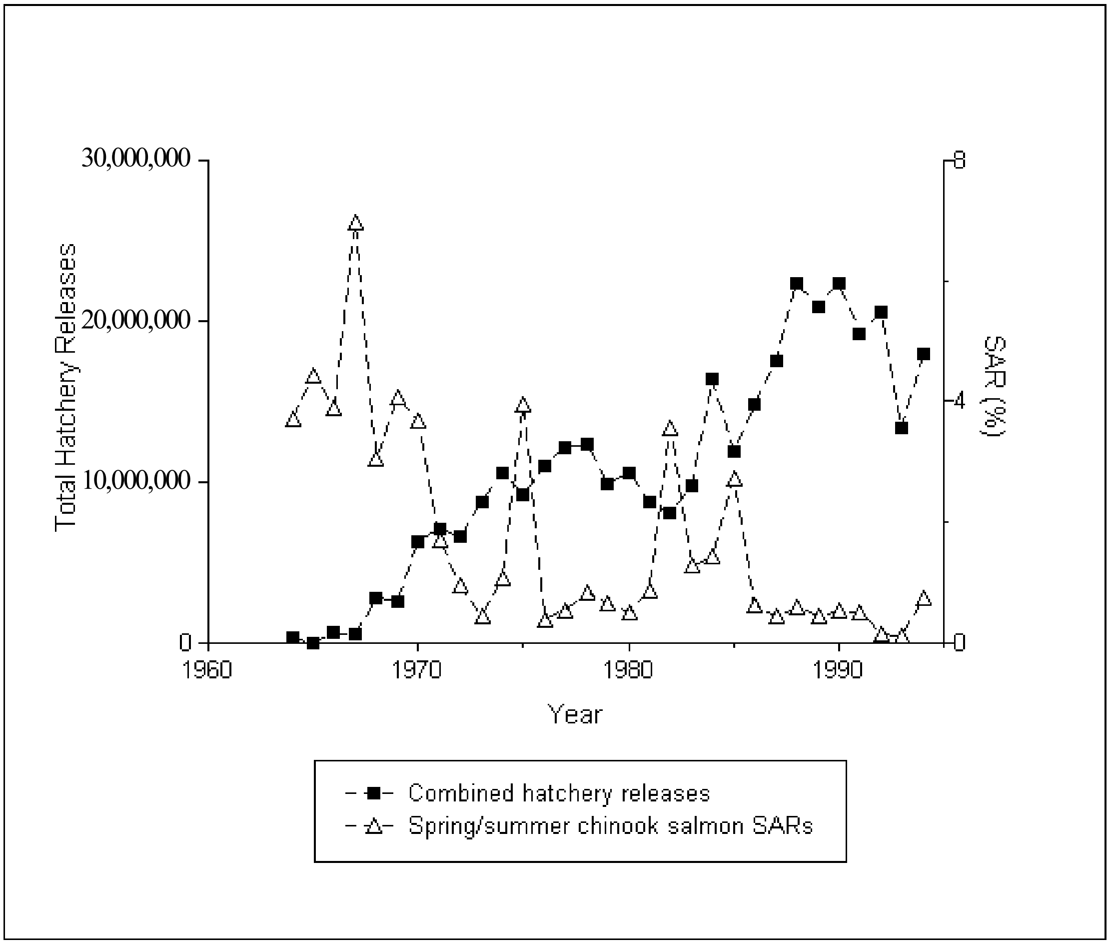


Note: Figure also shows the onset of low smolt-to-adult return rates (SARs) for wild spring/summer chinook salmon (Williams et al., 1998b). Smolt-to-adult return rates include escapement to the uppermost dam plus harvest.

Figure 2-2. Coincidence in Time of the Development of the Hydrosystem Cumulative Number of Mainstem (Lower Snake and Lower Columbia River) Dams

(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 passive integrated transponder (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 leftover mortality is unaccounted for and unexplained. However, the cause to which extra mortality should be ascribed remains elusive. Three major sources of extra mortality have been hypothesized: 1) hydropower system, 2) ocean regime shift, and 3) stock viability degradation. Each of these hypothesized sources of extra mortality is discussed below.

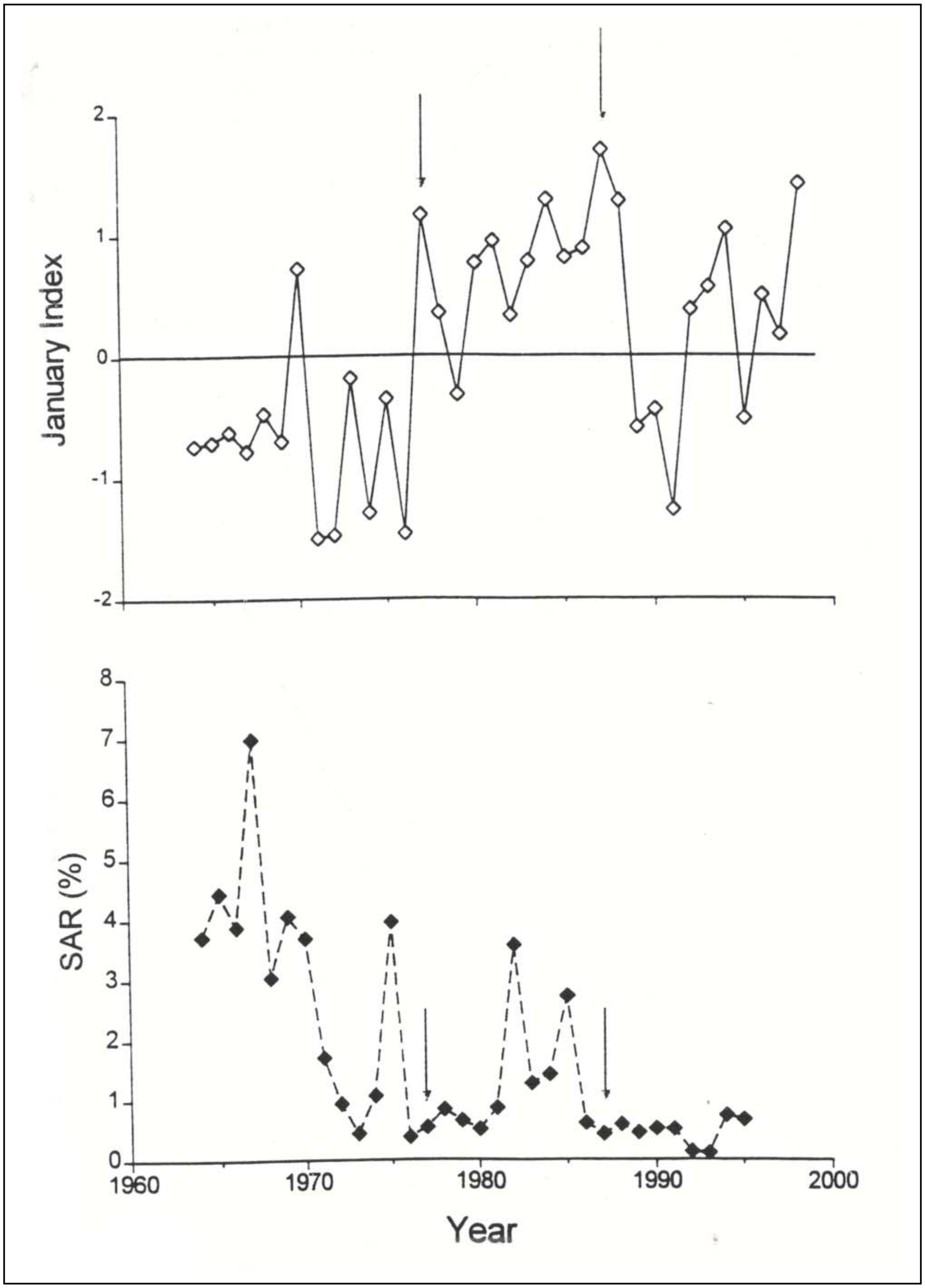


Note: Figure also shows the onset of low smolt-to-adult return rates (SAR) for wild spring/summer chinook salmon (Williams et al., 1998b). Smolt-to-adult return rates include escapement to the uppermost dam plus harvest.

Figure 2-3. Coincidence in Time of Hatchery Releases (Combined Releases of Spring/Summer Chinook Salmon and Steelhead; Williams et al., 1998a)

2.4.3.1 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 shared in common for 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 report (U.S. Fish and Wildlife Service [USFWS], 1998), and such dramatic changes may certainly have yielded a stock of fish less fit for life in the estuaries and oceans.



Note: Figure also shows 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 2 years when high values of the PDO coincided with low SARs. Estimates of the PDO index through March 1998 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 uppermost dam plus harvest.

Figure 2-4. Coincidence in Time of Anomalies in the Pacific Decadal Oscillation (PDO) Index

Table 2-1. Glossary of Frequently Used Technical Terms Page 1 of 2

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	Acronym for bacterial kidney disease, 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 rate	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, and subtracting out estimates of harvest and tributary harvest between the dams (see formula in Section 4.2.2).
CRiSP	Acronym for Columbia River Salmon Passage, 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-value	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 no different between transported and nontransported 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 nontransported fish).

Table 2-1. Glossary of Frequently Used Technical Terms. Page 2 of 2

Term	Definition
Extra mortality	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	Fish Leaving Under Several Hypotheses (FLUSH) is 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 3.4.3.2). The current ocean regime, and a shift on the timescale of years or decades, may affect the likelihood of recovery under any 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 [1998b] for a brief comparison).
Recovery	The process by which the ecosystem is restored so that 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 not meeting recovery and survival criteria, an action that succeeds in satisfying performance criteria over the widest range of assumptions.
Survival	The persistence of the species 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.

Although compelling data attributing mortality below Bonneville Dam to the hydropower system are not available, numerous data document how massively the hydropower system has altered the Columbia River (William et al., in press). Flow regimes have been altered from the natural processes. Habitats that are maintained by flooding and scouring and natural flow regimes are altered. Because the timing of fish migration is altered, fish experience different temperatures. Interactions with species are altered, and the reservoirs behind dams often harbor non-native species that prey on salmonid juveniles. Thus, although obstruction of fish passage and mortality while migrating through hydropower facilities are conspicuous and straightforward to measure, it is a mistake to think that these easily and directly observed impacts are the only impacts, or even the major impacts of dams. Many plausible mechanisms can be developed by which the presence of dams reduces fish fitness, and hence reduces survival below Bonneville or reproductive potential upon returning to the spawning grounds. The problem is that these extra mortality or reduced reproductive rates are very difficult to quantify experimentally. The ideal experiment (identical fish released in identical river systems with and without dams) is simply not a possibility. Consequently, less direct statistical analyses are the primary means of evaluating whether the hydropower system causes appreciable mortality or reduced fitness below Bonneville Dam.

2.4.3.2 Regime Shift Extra Mortality

A second important subset of extra mortality hypotheses is 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 from 1977 to 1986, coinciding with the onset of low smolt-to-adult return rates for salmon (see Figure 2-4 for a depiction of climate/stock performance correlations).

The linkage between ocean conditions and salmon performance is not simply a statistical correlation without a plausible mechanism; periods of positive anomalies for the PDO 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 adult spawners in the last few years (Welch et al., 2000). The marine survival of Oregon coastal coho salmon was 6.1 percent from 1960 to 1977, but only 0.6 percent 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 assumptions regarding how future ocean conditions will change. If ocean conditions are cycling, then salmon stocks will improve automatically without any management simply because the ocean condition 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.

It is important to realize that, although ocean conditions influence salmonid survival, poor ocean conditions are not sufficient to explain the extremely low smolt-to-adult returns for Snake River salmonids. An additional assumption is required, namely: the Snake River stocks are somehow more affected by poor ocean conditions than other stocks that are not experiencing such low smolt-to-adult returns. Some scientists discount this hypothesis because ocean cycles and fluctuations in the ocean environment have been a part of salmonid evolution for millennia, yet the stocks have thrived. Why then should ocean conditions now deplete the stocks so severely? This might 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 scarcity of data pertinent to these possibilities.

2.4.3.3 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). However, degraded stock viability is something of a catchall 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.

2.4.3.4 Assumptions About Mortality Below Bonneville Dam Determine Predicted Responses to Management Actions

Management could mitigate certain (but not all) causes of mortality below the Bonneville Dam. For example, if extra mortality is due to the fact that dams have dramatically altered river ecosystems (the 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 largely due to conditions in the ocean, then ocean factors outside the scope of this report will constrain management strategies, and actions such as dam breaching or habitat improvement may do little to recover the stocks.

2.4.4 Returning to the Natural River

The PATH process and NMFS-CRI approach 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 mechanism by which to analyze salmon survival and recovery. But there is 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 the system be moved?* The natural river is a multifaceted ideal. There are several ways to make a river look more natural. Which of the moves toward 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 is cut out? What are the essential features that get 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 is obtained through particular management actions that return the river closer to its natural state*?" 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. But ultimately, the currency for evaluating actions has to be salmon demography and population dynamics, not the physical attributes of a river alone.

3. The PATH Analytical Framework and Its Use by NMFS

3.1 Relationship Between PATH Process and NMFS Report

This anadromous fish assessment report is a product of the NMFS Northwest Fisheries Science Center (NWFSC). In developing this report, the syntheses and analyses conducted in the regional process known as the PATH was relied on extensively. 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 hydropower system 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's conclusions.

3.2 Logical Framework of PATH Analyses

It is difficult to develop simple management recipes 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. PATH 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, PATH breaks the salmon life cycle into stages and imposes a variety of assumptions on these stages about baseline conditions and likely changes due to different management actions. Historical data are used to narrow the range of assumptions and to 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.

PATH employed a formal decision analysis 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 even to increase to higher abundance levels [recovery])
- identifying the critical uncertainties that have the greatest impact on the predictions
- synthesizing the results and sensitivity analyses into summary statements about the biological merits of alternative management options.

It is useful to quote a PATH report (page 1 in Marmorek et al., 1998) to describe PATH objectives:

> *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:
>
> - 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)
> - 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
> - 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 uses a detailed life-cycle model to predict future chinook salmon populations under a variety of management alternatives. To implement the model, 8 to 10 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). Much work went into defining all of the critical assumptions and the uncertainties that underlie them. PATH is not, however, 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) are two of the strengths of PATH.

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 as 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, the SRP, and asked the panel 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., 1998), and 3) new data render some of the weighting obsolete. In particular, new data becoming available will allow alternative hypotheses to be rejected 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, as 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 should 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, PATH participants have fundamental disagreements about crucial hypotheses. Even though NMFS has participated in PATH, NMFS constantly updates its own scientific views as new information is obtained.

3.2.1 Developing Performance Measures

The performance measures used by PATH to judge the adequacy of the modeled alternatives were those used by NMFS along with nonquantitative 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 projections (i.e., modeled) requires some form of synthesis before the results are useful to fisheries managers. Therefore, each set of model runs was summarized relative to survival and recovery performance criteria.

The performance criterion used to assess the likelihood that a stock would survive is the requirement that in 70 percent of years the spawning abundance of a stock be above a certain low threshold. The specific threshold level depends on the characteristics of each stock and its natal stream (BRWG, 1994; Appendix D in Marmorek and Peters, 1998b). 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 percent of the years. PATH examined survival criteria for 24-year and 100-year timeframes.

Recovery performance was measured 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 (Biological Requirements Work Group [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 percent of the average spawner counts from before the 1971 brood year. To determine whether this recovery 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 target recovery level (i.e., 60 percent of the average NC-1971 brood year spawner counts).

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, 1995a). 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 percent of available populations (NMFS, 1995a). However, the level of 80 percent 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. Five index stocks would comprise 71 percent of the available populations, and six index stocks would comprise 86 percent. 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 that a simulation would satisfy NMFS’s survival criterion if six of the seven stocks were above a stock-specific threshold for at least 70 percent of the assumption sets. Similarly, the 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 for 50 percent of the assumption sets.

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 the four possible performance measures:

1) The 48-year recovery criterion provides the greatest distinction among management actions.
2) The 24-year survival criterion is the shortest time scale over which any quantitative analyses were performed. Thus, the survival criterion can help measure short-term risks.

One way of summarizing the myriad results from PATH is to simply calculate the average fraction of simulations that satisfy a survival or recovery criterion across all the assumption sets. The PATH documents refer to this as an “average probability” of meeting survival or recovery criteria (Marmorek et al., 1998). NMFS thinks it is important to avoid referring to these average fractions (or percentages) as probabilities because the definition of total probability space changes with each new assumption that is explored in the model. For example, 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. 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 percent for the 240 assumption sets could change to anything from 35 percent ([0.7 x 240 + 0 x 240]/2) to 85 percent ([0.7 x 240 + 1.0 x 240]/2). This is not a trivial point. These PATH probabilities do not translate in any way to a true probability (in the sense that we know the probability of getting 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 probabilities as defined by the PATH process do not represent the true probabilities intended when making a jeopardy decision. The PATH probabilities are useful for comparing the relative merits of different management options with respect to survival and recovery, but they are not literal probabilities regarding the fate of the populations. The predictions generated by the PATH analyses do not provide absolute predictions and should not be interpreted as such.

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 for 100 percent 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 percent 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.

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 was 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 they take 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 percent 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 corresponds 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, it is possible to examine how the Petrosky (1996) estimate of observation error affects the run reconstruction methodology. Deriso et al. (1996) found that a 25 percent 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 purpose for conducting the anadromous fish assessment is to summarize available biological information pertinent to the effects of the various Lower Snake River Hydropower Project 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 3-1):

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

A2) A1 plus maximize transportation (without surface collectors)

A2') A1 plus 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) inriver passage option (no transportation, no drawdown, flow augmentation as in A1, plus 123,400 hectare-meters [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 52,692 hectare-meters (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

Table 3-1. Hydrosystem Management Actions Examined by PATH

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 River and from the Dworshak Reservoir in the Clearwater Subasin.

Note: The A6 and A6' options have not yet been quantitatively defined. An "X" indicates the management action is implemented; a "--" indicates no action.

improvements). The effects of these management options were generally examined under a variety of scenarios (such as alternative harvest rates), as well as across a wide range of assumptions.

3.5 Limitations of the PATH Analytical Framework

There are several limitations of the PATH analytical framework that constrain NMFS' ability to draw on it for decision support. First, PATH analyses rely on a constellation of complicated models. It is difficult for any one person to run all of the models and generate results, or to quickly perform numerical experiments at the request of NMFS or other managers. Although PATH participants have been responsive to NMFS' requests for analyses, the long response time limits the scenarios that can be examined. In addition to making analyses slow, the large number of assumptions and parameters makes the PATH modeling framework something of a "black box," which is too intricate to be understood to the public. For example, there are hundreds of parameters that represent constants or rates in any one PATH simulation run.

A second limitation of PATH is that no populations in any simulation run, regardless of the scenarios or assumptions employed, has ever fallen as low as one spawner over a 100-year time period. For example, even in the scenario where no dams are breached, ocean conditions do not markedly improve, and no further management is taken to improve salmonid stocks, none of the PATH simulations show any stocks going extinct within 100 years. This clearly does not reflect the extinction risk expected given that some of these stocks have critically low populations and have recently exhibited downward trends. While PATH examines model output in terms of "survival standards," it does not afford an assessment of extinction risk. Yet risk of extinction is one of the most important risks to calculate in any population viability analysis.

Third, the PATH analytical framework does not lend itself to examining the consequences of risk factors beyond the hydropower system. For example, PATH does not examine improved habitat conditions, reduced predation due to hatchery modifications, or completely curtailed harvest, and there is a general lack of integration of all possible management actions. There is a need to examine a broader menu of management interventions to assess strategies for salmonid recovery.

Finally, there are certain technical constructs of PATH, most notably differential delayed transportation mortality (D-values to be discussed in detail in later sections) and extra mortality that lead to a great deal of confusion. Uncertainty is better cast in terms of easily understood quantities that could potentially be directly measured, not in terms of derived parameters such as extra mortality or differential delayed transportation mortality.

In summary, while the PATH framework has succeeded in building detailed mechanistic models that analyze the intricacies of fish passage and alterations of the hydropower system, the details detract from providing a clear picture when looking beyond the hydropower system for salmonid recovery. This key limitation, as well as the other limitations discussed in this section, prompted NMFS to undertake a complementary analytical framework beginning in June 1999. This new framework, CRI, is still under development, but preliminary results from it appear in Section 8 of this report.

4. PATH Analyses of Spring/Summer Chinook Salmon

4.1 Population Ecology and Trends

The Snake River Basin includes an area of approximately 277,130 square kilometers (107,000 square miles), almost one half 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 4-1), indicating that some populations are subject to a high extinction risk (in Section 8 probabilities of extinction are calculated for these stocks).

4.1.1 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 1,442 kilometers (896 miles) from the sea.

The Snake River was probably the major producer of spring/summer chinook salmon in the Columbia River Basin, producing about 39 percent of the spring chinook and 45 percent of the total summer chinook salmon run at one time (Mallett, 1974). The estimated total production of the Snake River probably exceeded 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 percent of the spring/summer chinook salmon entering the Columbia River from 1957 to 1960 (Fulton, 1968). Adult escapement to the Snake River averaged about 37,100 spring chinook and 22,300 summer chinook from 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 withdrawals, 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

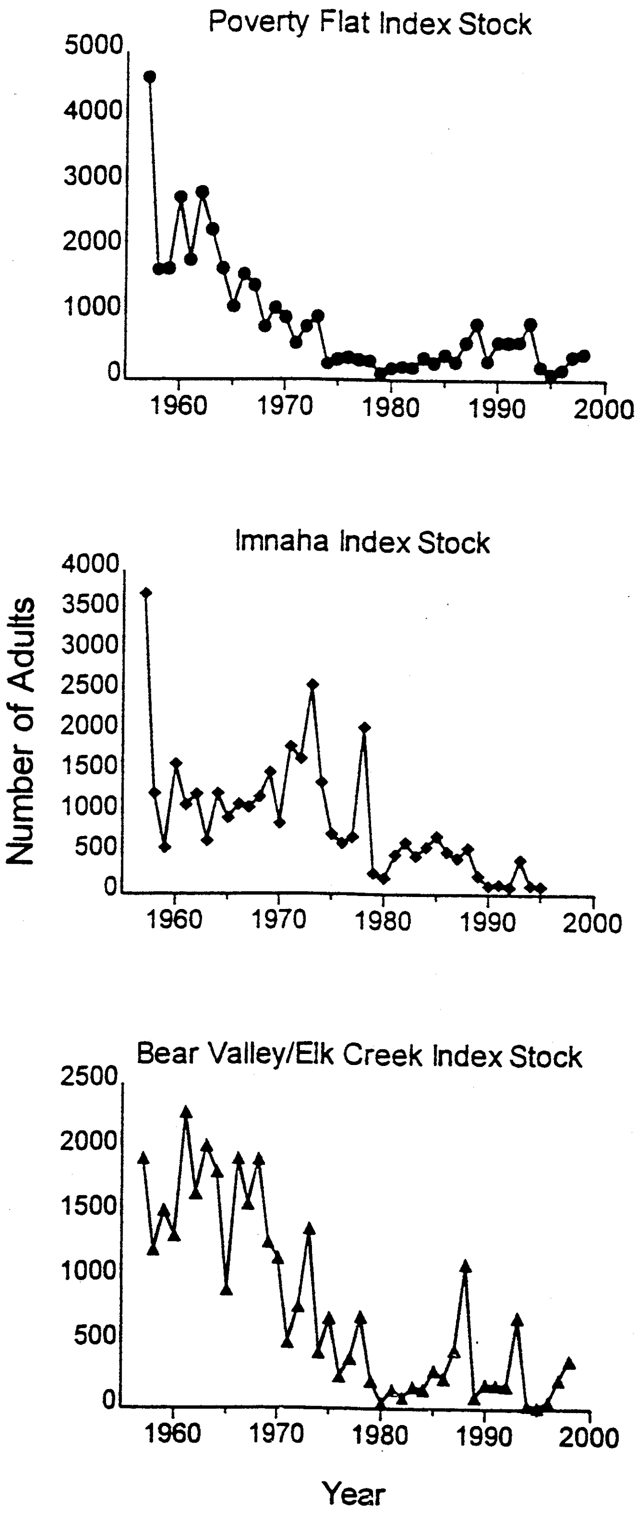


Figure 4-1. Declining Trends in Adult Returns for Three of the Spring/Summer Chinook Index Stocks Modeled by PATH (Poverty Flat, Imnaha, and Bear Valley/Elk Creek)

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 creating channels in riparian sloughs and tributaries. In addition to eliminating habitat, these activities (as well as mining and industry) have decreased the 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 substantially 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 percent from historical levels by the 1960s (Table 4-1).

Table 4-1. Changes in the Number of Spawning Stream-Type Chinook Salmon and Contribution of Biomass, Nitrogen and Phosphorus from Their Carcasses

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

Note: These data are for 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 percent and phosphorus 0.36 percent of wet body weight in Pacific salmon (Larkin and Slaney, 1997).

4.1.2 Hatchery Production

The production of salmonid smolts from Snake River hatcheries (both of spring/summer chinook and steelhead) has increased greatly 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 2-3). Based on the coincidence of these factors in time, NMFS is exploring the possibility 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, before arrival at the first Snake River dam, and were probably exacerbated in areas where fish concentrate (forebays, bypass systems, collection raceways, and barges). The effects of hatcheries may be greater for Snake River stocks than for mid-Columbia River stocks for the following reasons:

- The migration corridor before 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 Snake River stock.
- 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.

Within the context of PATH analyses, interactions with hatchery fish are one possible source of extra mortality (and are placed in the category of reduced stock viability).

4.2 Adult Harvest and Upstream Passage

4.2.1 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/summer chinook salmon in mainstem fisheries is estimated to have ranged from 3 to 8 percent since 1978 (Marmorek et al., 1998).

4.2.2 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.

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

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 from 1977 to 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 before 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.

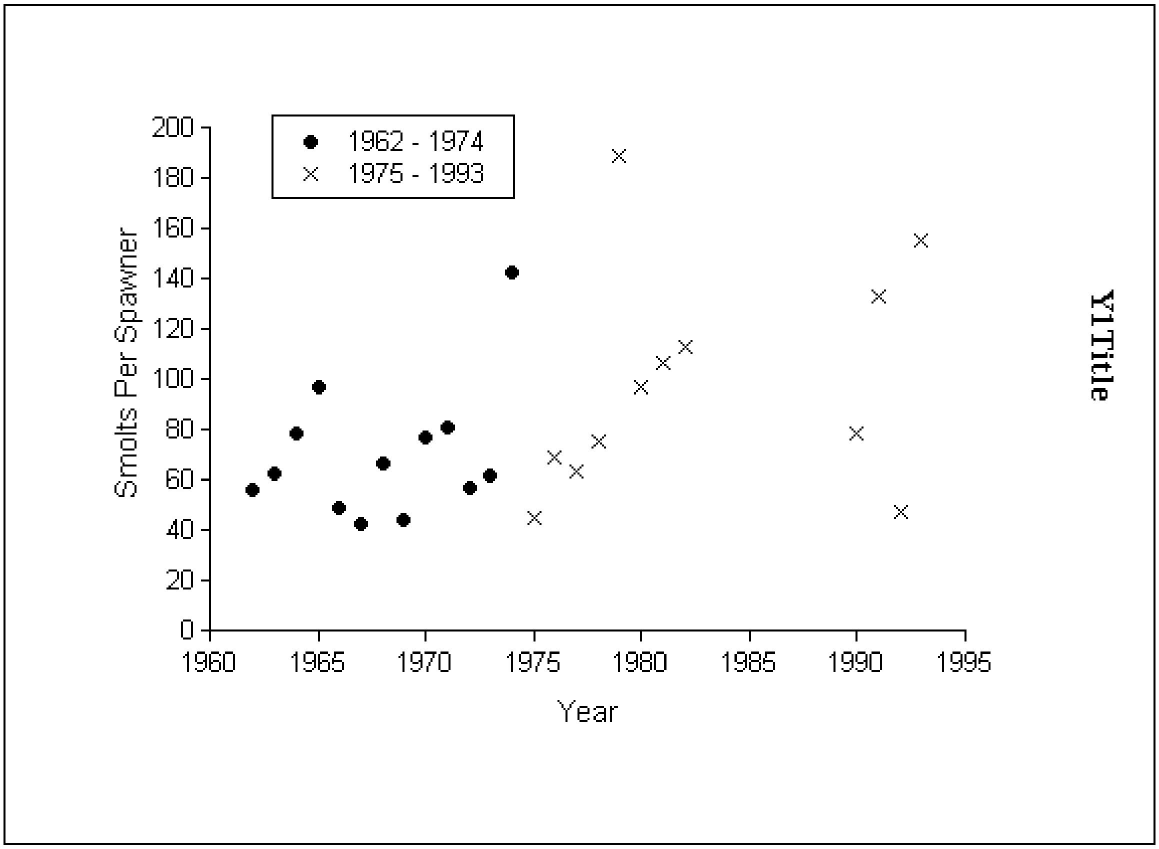
The conversion rate method of estimating upstream passage survival has a potential bias related to the differential fallback of upstream migrating adults at the dams where counts are made. A detailed discussion of this potential problem, including a comparison of upstream survival estimates made using different methods, is included in Section 5.2.2. However, for spring/summer chinook salmon, survival estimates derived from PIT-tag experiments (C. Paulsen, memorandum, February 17, 1999) were similar to estimates based on conversion rates. In addition, the radio-telemetry studies summarized in Marmorek et al. (1998) indicate a mean project survival estimate for the four-dam Snake River reach of 0.847, essentially identical to the conversion-rate based estimate of 0.85 for the same reach.

4.3 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 knowledge of the relationship between quantity and quality of habitat and fishery productivity is imperfect.

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 2-4), there was no concordant precipitous decline in habitat productivity as measured by smolts per spawner (Figure 4-2).

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



Note: 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 4-2. Number of Spring/Summer Chinook Salmon Smolts per Spawner (Collected Above Lower Granite Dam; from Petrosky and Schaller, 1996)

habitat conditions (Marmorek et al., 1996). However, 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).

4.4 Smolt-to-Adult Life Stage

Estimates of smolt-to-adult return (SAR) rates (Figure 2-4; lower graph) indicate that survival has dramatically declined over the last 30 years (Marmorek et al., 1998; 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
- 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 4.2.2). Thus, the major unknown factor is the cause of mortality in the estuary and ocean. The PATH analyses break estuary and ocean mortality into two major categories:

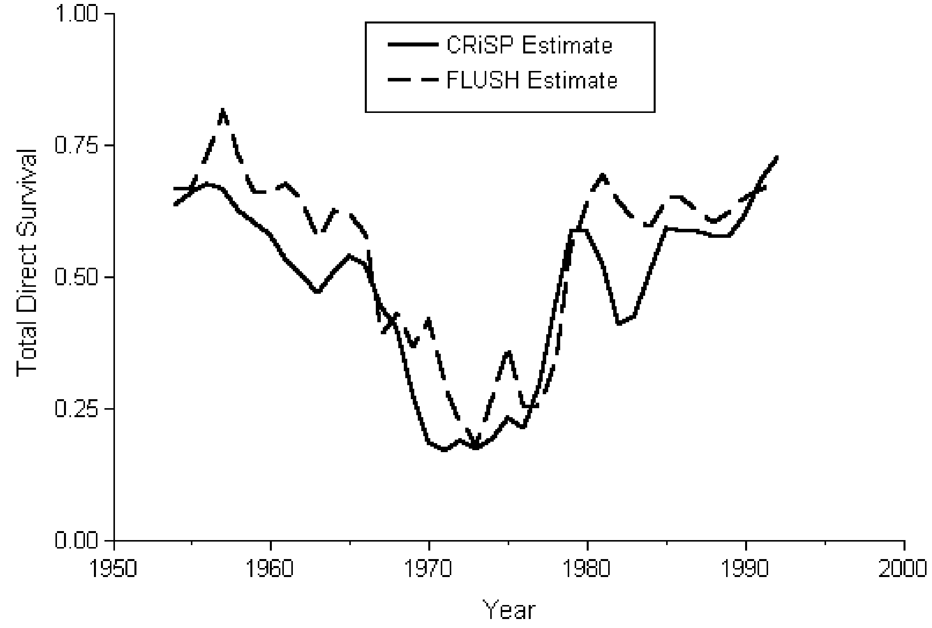
- differential delayed transportation mortality, which is experienced only by transported fish
- extra mortality, or the unexplained mortality affecting Snake River stocks below Bonneville.

4.4.1 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 uppermost Snake River facilities (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. At this point, however, detection rates at Bonneville Dam are relatively low and trawl-mounted PIT-tag detectors are still in the 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 independent estimates of passage survival at some mainstem dams. 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 passage models (CRiSP versus FLUSH; see glossary in Table 2-1). For a full discussion of the differences between these models, consult the PATH reports for fiscal years 1997 and 1998 (Marmorek and Peters, 1998b; Marmorek et al., 1998). The passage models estimate survival of the total population of fish from the head of Lower Granite Reservoir to the tailrace of Bonneville Dam. 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 the 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 believes that, if FLUSH and CRiSP were forced to run with identical D-values, the models would generate very similar predictions.

Biologically, the important point about spring/summer chinook salmon direct survival is captured in Figure 4-3. Direct survival to below Bonneville Dam declined sharply in the late 1960s and early 1970s. This decline in migration survival parallels the decline in SARs and the collapse of



Note: Survival rates are graphed as 5-year moving averages. Direct survival does not account for any delayed mortality of either transported or inriver migrants.

Figure 4-3. Total Direct Survival (Transported Plus In-River Migrants) of Juvenile Spring/Summer Chinook Salmon to Below Bonneville Dam

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 4-3). However, SARs 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.

4.4.2 Accounting For Climate Effects in Smolt-to-Adult Return Rates

Before examining hydropower system effects in terms of depressed SARs, the influence of climate and ocean conditions has to be factored out.

Survival through the estuary and ocean life-history phase is affected by year-to-year variation and multiyear 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 SARs.

The effect 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., 1998). Instead of assuming one particular link 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 4.5.1.4.

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. (1996). 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., 1996; 1998). 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 and 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., 2000). 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 strikingly low SARs of 1992 and 1993 for spring/summer chinook salmon. However, a similar ocean-based survival for spring/summer chinook salmon as for coho salmon and steelhead cannot necessarily be inferred because it is not known whether the species occupy similar ocean habitats.

An important source of uncertainty about ocean conditions arises when considering options for simulating the future. For example, when simulating possible future 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. The PATH 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 rates of simulated populations are optimistic based on ocean effects.

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

4.4.3.1 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 by PATH to date.

NMFS used data derived from wild fish PIT-tagged as juveniles above Lower Granite Dam from 1994 to 1996 to derive estimates of *D*. 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: PIT-tagged fish first detected and transported from Lower Granite, Little Goose, and Lower Monumental Dams represented transported nontagged fish from the same location. Data on transported fish from McNary Dam in 1994 were not used, as it appeared problems existed with the transportation system, transportation was not implemented there in 1995 or 1996, and under the four-dam drawdown scenario, transportation from McNary Dam is not envisioned. However, because most nontagged 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 to Bonneville Dam passed the dams mainly via spill and turbine routes. Thus, PIT-tagged fish detected (bypassed) multiple times were not 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, in press). NMFS used these methods to estimate the number of PIT-tagged juvenile fish that survived to the tailrace of Bonneville Dam and that used passage routes representative of nontagged downstream migrant fish. In 1994, nearly all nontagged fish that entered bypass systems at Lower Granite, Little Goose, Lower Monumental, and McNary dams were transported. Thus, the PIT-tagged fish that best represented the nontagged fish that survived to Bonneville Dam were those in the "never-detected" group. During the 1995 and 1996 migrations, however, the collection system at McNary Dam operated in "full bypass mode," returning all fish (tagged or nontagged) that entered the bypass system to the river. Thus, for 1995 and 1996, 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.

From 1994 through 1996, the combined adult returns in any one year of wild spring/summer chinook salmon juveniles PIT-tagged above Lower Granite Dam and either transported or migrated downstream in the hydropower system ranged from only 6 to 21 fish. Thus, the estimates of *D* derived from these data have low precision. Estimates of *D* for the three years (with boot-strapped 95 percent confidence) were 0.9 (0.0 to 1.8), 0.6 (0.0 to 1.1), and 1.0 [(0.6) to 2.6], respectively. Pooling the estimates for the three years provides a *D* estimate of 0.8 (0.3 to 1.3).

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 percent 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 is sampled). Because both perspectives have merit, this report presents results for a range of D-values.

One review of this appendix (Schaller et al., 1999) makes the argument that NMFS' calculation of $D = 0.81$ is in error. Essentially the argument presented is that $D = 0.81$ is an outlier in a frequency distribution of different possible D-values, with each value corresponding to a different way of calculating *D*. NMFS finds this argument fallacious. There are only a few (2 to 5) reasonable ways to calculate *D*, not 112 different ways. Science does not proceed by identifying all possible ways, for example, of calculating a planet's orbit or a mutation rate; instead there are some methods that are better than others. This is true for calculations of *D* as well. In particular, while the concept of differential post-Bonneville survival for transported and inriver fish is general, the parameter *D* has a specific meaning, given by the manner in which it is applied in the PATH life-cycle models. In this model, *D* is defined as the ratio of two parameters: λ_T, the post-Bonneville survival for transported fish, and λ_C, the post-Bonneville survival for fish that arrive below Bonneville via inriver routes. In particular, the traditional "T:C" ratio of Lower Granite smolt-to-Lower Granite adult return rates for the two groups can be expressed as the product of the ratio of juvenile survival from Lower Granite Dam to Bonneville Dam and the ratio of post-Bonneville Dam survival:

$$T:C = \frac{SAR_T}{SAR_C} = \frac{V_T \lambda_T}{V_C \lambda_C} = \frac{V_T}{V_C} D$$

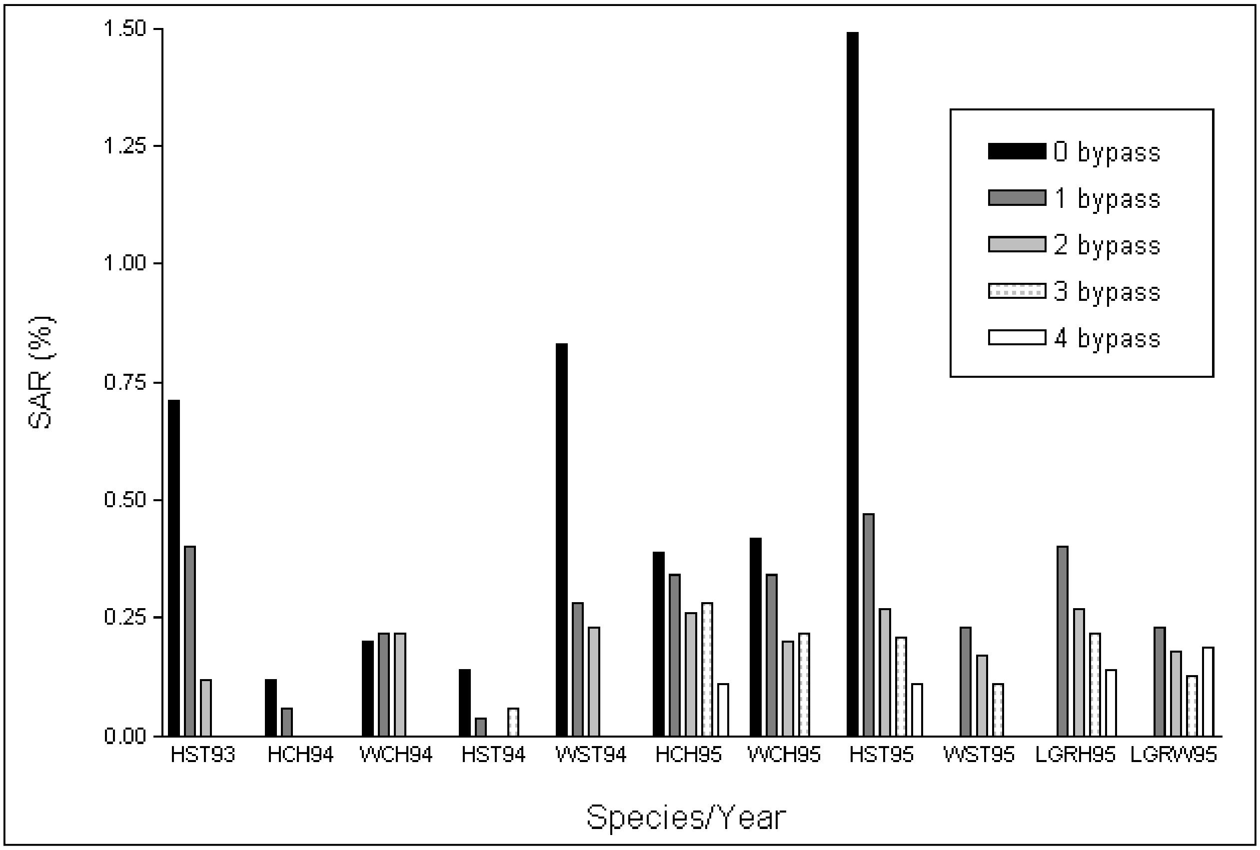
The PATH life-cycle models assign the same value of λ_T, and hence *D*, to all transported fish, regardless of the dam from which they were transported. Thus, if post-Bonneville survival does vary depending on transport site, the PATH *D* is actually a weighted average of the differential mortality for the various transport sites included in a particular prospective scenario.

Moreover, all previous PATH analyses (non PIT-tag) that attempted to estimate *D* were based on transport studies that transported fish from Lower Granite or Little Goose dams. The resulting estimated *D-values* have then been applied to all transported fish in the PATH models. In NMFS' analysis, the choice to use fish transported from Lower Granite, Little Goose, and Lower Monumental dams was made because most prospective scenarios involving transportation place heavy emphasis on collecting and transporting fish at the upper dams.

When using data from PIT-tagged fish to estimate parameters for the PATH models, it is important to remember that those models are intended to represent the runs at large, and that PIT-tagged fish are not necessarily representative of nontagged fish in every regard. Especially important in the case of estimating *D* is the fact that the proportions of PIT-tagged fish that experience certain detection histories is vastly different from the proportions of nontagged fish. It was this realization that led to the use of “never detected” PIT-tagged fish as the most proper group to use to represent nontagged fish that remain in the river. PIT-tagged fish that entered collection systems in 1994-1996 were usually returned to the river; nontagged fish in collection systems were transported. (The situation changed beginning in 1997 when many PIT-tagged hatchery fish were purposefully transported from Lower Granite Dam for the Idaho Hatchery PIT-Tag Study.) Thus, of the fish that remained in the river and survived to Bonneville Dam, a much higher proportion of PIT-tagged fish experienced one or more bypass systems than did their nontagged counterparts.

The same care must be taken to define the group of transported PIT-tagged fish that is to represent transported nontagged fish to estimate *D* for the PATH models. Most PIT-tagged fish were returned to the river at Lower Granite and Little Goose dams. The result is that, comparing transported PIT-tagged and transported nontagged fish, a higher proportion of PIT-tagged fish were transported from lower dams than their nontagged counterparts. To say it another way, nontagged fish were transported the first time they were bypassed; more PIT-tagged fish were returned to the river and vulnerable to transportation at lower dams. Estimates of *D* based on PIT-tag data must account for this bias toward lower-river transport among PIT-tagged fish.

Annexes B, C, and D provide detailed discussions of how NMFS estimates passage survival using PIT-tag data, and in turn calculates *D*. It is clear that the calculation of *D* is not simple, because it must contend with different passage routes and sources of mortality. Nonetheless, the statement “more data are unlikely to perfect our understanding of *D* or eliminate the uncertainty” (Schaller et al., 1999) does not seem to be a reasonable conclusion. This is a challenging scientific problem, but that does not mean that more data and experiments cannot reduce uncertainty. It is hard to imagine a science for which more data and experiments will not teach us anything. For example, by quantifying smolt-to-adult returns for PIT-tagged fish that experience different bypass histories, it may be possible to refine our understanding of the impacts of hydroprojects on survival (Figure 4-4).



Note: These rates depend 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 4-4. Estimated Smolt-to-Adult Return Rates (Percent)

Also, the entire *D* debate may represent a problem with how the question is posed. Essentially the real question is: do the hydropower systems and transportation systems somehow reduce the fitness of fish? Consequently, it may be fruitful to look directly for evidence of fitness reductions by following individual fish.

4.4.4 Extra Mortality

Extra mortality is defined as any mortality of Snake River salmon and steelhead that occurs outside of the juvenile migration corridor and that is not accounted for by productivity parameters in spawner-recruit relationships, estimates of direct mortality within the migration corridor (from passage models), differential delayed transportation mortality, or common-year climate effects influencing both Snake River and Lower Columbia River stocks (Marmorek et al., 1998). In the context of PATH, extra mortality was estimated as any mortality not accounted for by other terms in the life-cycle model (see Annex 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 were hypothesized to have contributed to this mortality. These included:

- climate/environmental trends specifically affecting Snake River salmon runs (or having a greater impact on Snake River Salmon runs than on mid- and lower-Columbia runs)
- effects of factors other than climate and other than the Snake River dams (generally referred to as declines in stock viability)
- delayed effects of hydrosystem passage (not encapsulated in differential delayed transportation mortality).

4.4.4.1 Climate Regime Shift Hypothesis

A long-term, cyclical shift in climate regime over 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 around brood year 1975. The climate regime is hypothesized to return to an above-average (favorable) condition starting with brood year 2005. If a regime-shift caused extra mortality it 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. The SRP for PATH felt that there was little evidence for the regime shift extra mortality hypothesis (Weight of Evidence Report, Marmorek and Peters, 1998b).

4.4.4.2 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 BKD within the wild population. As a result, it was hypothesized that the mortality of juvenile fish increased after they exited the hydrosystem as compared to mortality observed in earlier years.

More recently, a wide variety of biological mechanisms have been hypothesized as causes of reduced stock viability. For instance, hatchery releases may negatively impact wild Snake River chinook salmon directly (predation) or by subtly elevating stress levels. 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 substantial predation on spring/summer chinook salmon smolts. The increases in hatchery production were instituted primarily 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 are 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 to 30 million smolts annually (albeit mainly hatchery steelhead smolts and not chinook smolts). These terns were not present in the estuary before 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).

4.4.4.3 Hydropower Hypotheses Regarding Extra Mortality

The most obvious extra mortality hypothesis involves the hydropower system itself. Clearly the dams on the Snake River dramatically altered this ecosystem (see USFWS Coordination Act Report). 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 facilities. 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 percent) and hence 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. Obtaining direct data to support this hypothesis is not easy.

4.5 Analysis of Hydrosystem Management Alternatives

4.5.1 Future Effects of the Hydrosystem Management Actions

The PATH process, using each of the two alternative passage models, CRiSP and FLUSH, projects 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 previously. The results are expressed as a series of adjusted inriver survival values for use in the life-cycle analyses described previously.

For completeness, a large number of assumptions and modeling details are outlined in this section, giving the impression of a very complicated story. However, the bottom-line message is straightforward. In particular, assumptions about extra mortality and differential delayed transportation mortality ultimately determine the results to a great extent.

4.5.1.1 Assumptions Used in Simulations of Future Conditions

Inriver Survival

Using the passage models, projected survival rates for inriver 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 inriver survival: dam passage elements and reservoir passage/survival studies. The same elements used in assessing retrospective passage survivals were incorporated into the prospective modeling. Spill levels were set depending on the particular future management option being assessed. Spill survival was assumed at 98 percent. 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 4-2).

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 percent reduction in reservoir mortality. These two values were chosen to represent the extreme bounds for probable effectiveness of predator removal.

Table 4-2. Set of Assumptions and Alternative Values for These Assumptions, Used in the PATH Analyses

Page 1 of 2

Uncertainty	Hypothesis Label	Description
Uncertainties/hypotheses related to downstream passage to Bonneville Dam		
Inriver survival assumptions—Passage models	PMOD1	CRiSP direct survival estimates.
	PMOD2	FLUSH direct survival estimates.
Fish guidance efficiency (FGE)	FGE1	FGE w/ELBS > FGE w/STS. (ELBS = extended-length submerged bar screen). (STS = submerged traveling screen).
	FGE2	FGE w/ELBS = 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 percent reduction in reservoir mortality resulting from predator removal program.
	PREM3	25 percent reduction in reservoir mortality.
Duration of preremoval 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.

Table 4-2. Set of Assumptions and Alternative Values for These Assumptions, Used in the PATH Analyses. Page 2 of 2

Uncertainty	Hypothesis Label	Description
Other uncertainties/alternative hypotheses		
Transportation models	TRANS1 or T1 (FLUSH only)	Relationship established between TCR and FLUSH inriver survival, based on data from all transport studies conducted at LGR and LGO dams from 1971 to 1989. This relationship, and FLUSH inriver 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*)
	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 1970s and associated CRiSP inriver survival rate estimates. Post-1980 retrospective analyses use average D-value estimated from four T:C studies in 1980s, and CRiSP inriver survivals. For prospective analyses, D-value randomly selected from four 1980 values.
Distribution of extra mortality	ALPHA	Extra mortality is specific to each subregion, 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.

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 inriver 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 among 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 percent.

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 nontransported 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. (1998). For the purpose of this report, it is important to note only 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 near-natural river drawdown (breaching) of four Snake River mainstem reservoirs (Lower Granite, Little Goose, Lower Monumental, and Ice Harbor Dams). The second alternative (B1) involves a combination of near-natural river drawdown of John Day Dam on the mainstem Columbia River with the four-reservoir 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 beginning when fish populations equilibrate to the end of the simulation period.

For each period, the PATH process requires assumptions about the duration of these four periods and estimates of the adult and juvenile survival rates that are expected (Table 4-3). The potential for increased juvenile mortality associated with the transition following drawdown was considered in a set of PATH sensitivity analyses (Marmorek et al., 1998). Two scenarios were considered: decreasing inriver survival for the first 5 years after drawdown by 10 percent and decreasing in-river survival for the first 5 years after drawdown by 50 percent. The 10 percent and 50 percent 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 4-3, the removal effects from breaching do not include any impacts on juvenile or adult survival; the general types of effects that might occur for all salmonids are discussed 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 facilities) 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.

Table 4-3. 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/]
Preremoval	3 years or 8 years	Determined by passage models	Current estimates
Removal	2 years	No change from preremoval period	No change from preremoval
Transition	2 years or 10 years	Linear increase from preremoval survival to equilibrated survival	Linear increase from preremoval 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 facility reaches.

2/ Conversion rates

The transition period is defined as that time between the end of the construction period and the period when the near-natural 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 alongside 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 near-natural 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 the following three processes:

- response of predator populations to the change from reservoir to near-natural 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
    - changes in channel morphology and microhabitat distribution may affect distribution of predators and juvenile chinook salmon, which would affect encounter rates
- decreased fish travel times that result from increased water velocities reducing exposure of juvenile chinook salmon to predation
- 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 enough to counteract the positive effects of reduced travel times. A very limited amount of information is available on predator densities and predation rates in near-natural 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 near-natural Snake River is tenuous, and the data for making broad conclusions are sparse. 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 inriver passage assumptions on expected survival under the alternative lower 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 inriver and transported migrants. Both CRiSP and FLUSH models project relatively high estimates of total direct survival for the future under the Existing Conditions Alternative (A-1) and the Maximum Transport of Juvenile Salmon Alternative (A-2), reflecting the high proportions of the run transported. The projected estimates of direct total survival to below

Bonneville Dam for Alternatives A-1 and A-2 exceed the corresponding juvenile survivals projected for the Dam Breaching Alternative (A-3) under both modeling systems.

Estimates of system survival for inriver 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.

4.5.1.2 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 Annex A 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 4.1. 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 2.2.1.

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 Annex A to this report and more fully in Appendix A.3.2 in Marmorek and Peters (1998a). 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 (Marmorek et. al., 1988).

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 survival estimates were drawn based on those probabilities in the prospective model runs.

4.5.1.3 Results of the Decision Analysis

The results of the PATH analytical work conducted to date have been summarized in a series of reports. 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, 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, the results of alternative hydrosystem actions can be compared 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 fraction of runs that exceeded the survival threshold or recovery levels under that set of assumptions. To incorporate the effect of uncertainties, PATH used 4,000 100-year replicate Monte Carlo simulations for each set of assumptions. In Table 4-4, the average fraction of runs that exceeded these escapement levels is summarized for each of six alternative management actions.

Table 4-4. Average Fraction of Runs (Across All, Equally Weighted Assumption Sets) Exceeding Survival and Recovery Escapement Levels for Spring/Summer Chinook Salmon for Alternatives A1, A2, A2', A3, and B1

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)

Note: 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.

Table 4-4 indicates clearly that dam breaching (either A3 option) averages an 82 percent frequency of meeting recovery population escapement criteria, whereas no-breaching averages a 47 to 50 percent frequency of meeting the recovery criteria. Thus, breaching provides an additional

30 percent 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). The difference between dam breaching and transportation is even more dramatic if one asks over what fraction of assumption sets are recovery criteria satisfied. Breaching doubles the fraction of assumption sets that end up with recovery (from at most 47 percent to 100 percent, Table 2.2.4.3 of Marmorek et al., 1998).

One problem with reducing the analysis to a single number for each management action (the average fractions shown in Table 4-4), 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 fractions associated with the middle 50 percent of outcomes. An approximation of the jeopardy criterion NMFS used in the 1995 FCRPS Biological Opinion is indicated by the dashed horizontal line across each graph (70 percent for survival criteria; 50 percent for recovery criteria).

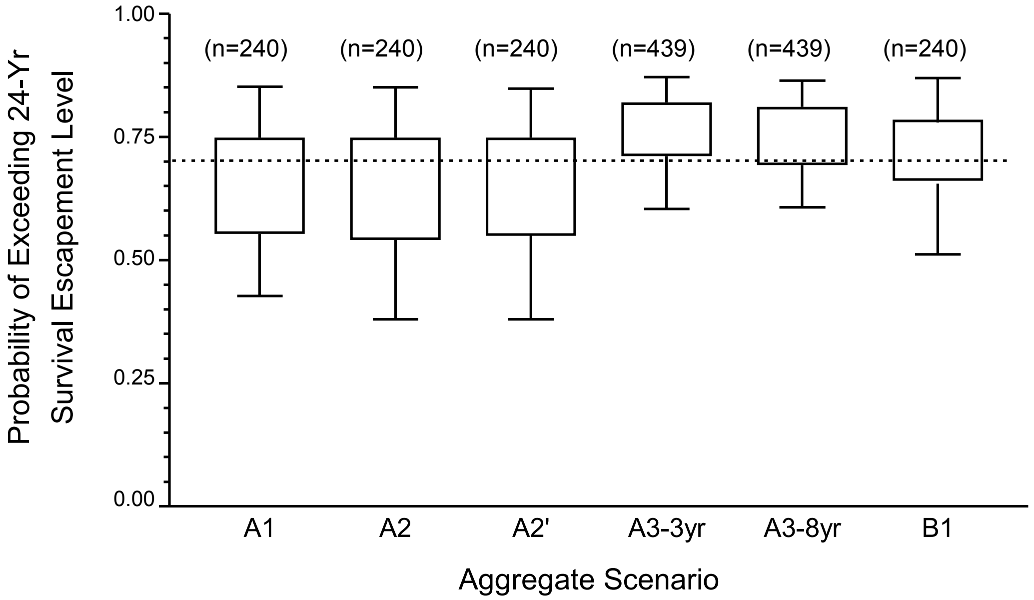
The ability to meet the 24-year survival criterion (Figure 4-5) is strongly related to the current status of the stocks, although alternative management actions have some effect on the projected results. In general, the actions involving drawdown of dams result in higher projected frequencies of meeting 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 probably optimistic.

The 48-year projections of performance relative to the recovery criterion (Figure 4-6) give the greatest contrast among the alternative hydrosystem actions. Almost all actions involving Snake River drawdown are projected to exceed the 50 percent recovery performance criteria, on average. In dramatic contrast, A1, A2, and A2' (no drawdown options) fail to meet the recovery criterion in most of the runs. In addition, the size of the middle 50 percent box for dam breaching is consistently smaller than the middle 50 percent 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 percent boxes).

4.5.1.4 The Key Assumptions Underlying Critical Comparisons for Decision Making

The results summarized in Figures 4-5 and 4-6 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 quantification of the effects of particular assumptions and thereby identifies 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., 1998). To illustrate this graphically, NMFS has focused on the contrast between A1 (current operations, no breaching) and A3 (dams breached in

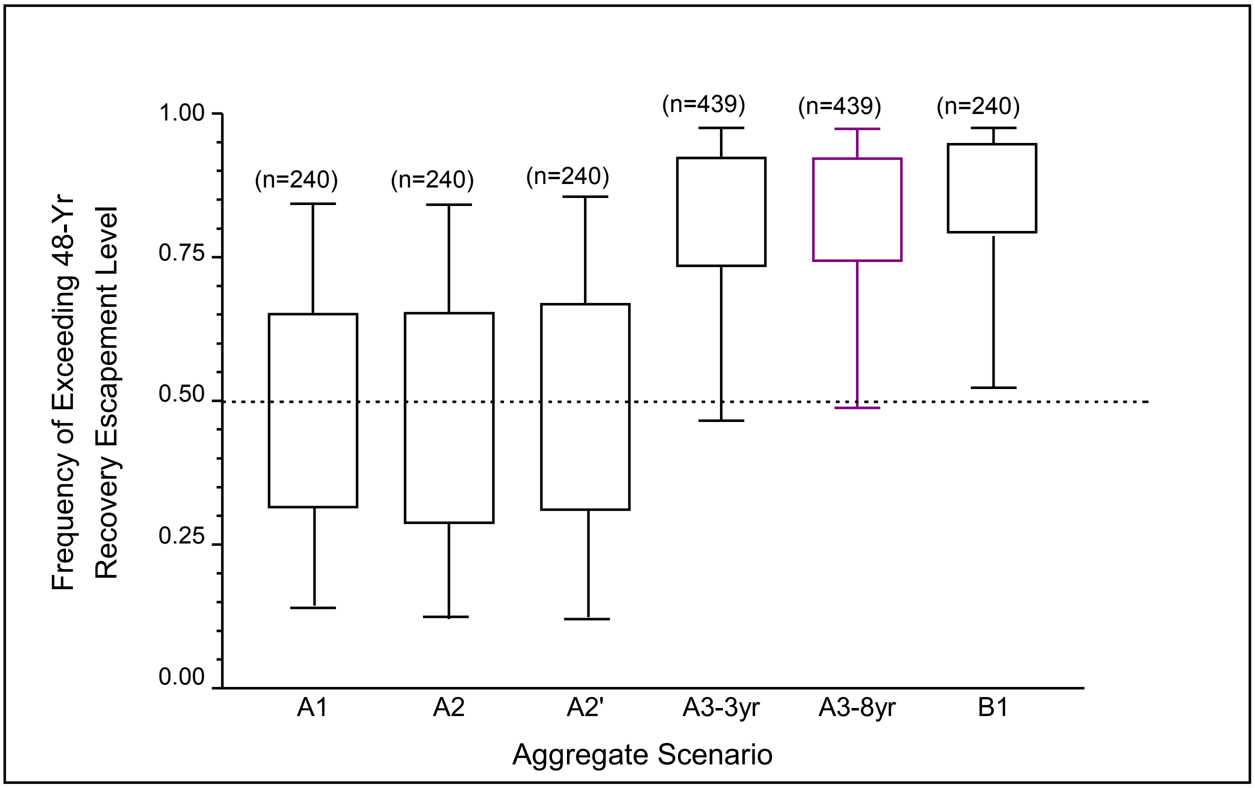


Note: Data are calculated according to the PATH prospective life-cycle model. Alternative A3 (drawdown) was evaluated assuming both 3-year and 8-year delays. "n" indicates the number of assumption sets for each scenario. Dashed line indicates the 24-year survival criterion. See text for explanation of "Box and Whisker" plots.

Figure 4-5. Frequency of Exceeding the 24-Year Survival Escapement Level for Spring/Summer Chinook Salmon under Alternatives A1, A2, A2', A3, and B1

3 years) and examined how the frequency of exceeding recovery criteria depends on these critical assumptions (Figure 4-7).

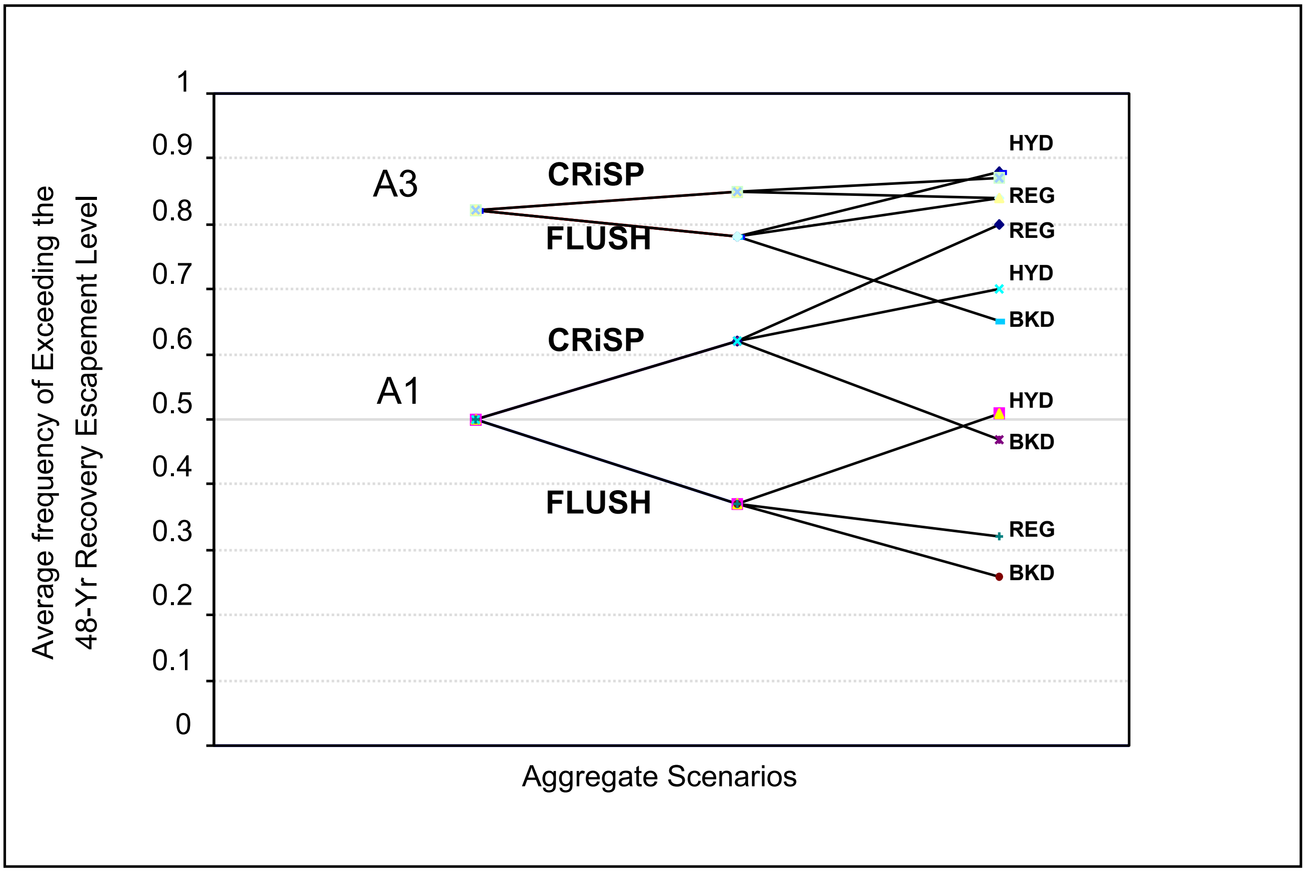
In light of recent PIT-tag data suggesting that D-values may be higher than have been used on average in the PATH simulations (see Section 4.4), NMFS ran a series of prospective simulations to examine the effect of higher D-values (and hence lower differential delayed transportation mortality) on the frequency with which the 48-year recovery criterion is expected to be met. The results of these runs, shown in Figure 4-8, 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 percent for dam breaching. This would still represent a substantial reduction in risk (64 percent frequency of meeting the 48-year recovery criterion versus 53 percent), but nowhere as dramatic as the 30 percent difference 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 4-9. If D= 0.8, breaching may still yield a dramatic reduction in risk (19 percent), but only if extra mortality is due to the hydrosystem. Indeed, with $D = 0.8$, if extra



Note: Data are calculated 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 4-6. Frequency of Exceeding the 48-Year Recovery Escapement Level for Spring/Summer Chinook Salmon Under Alternatives A1, A2, A2', A3, and B1

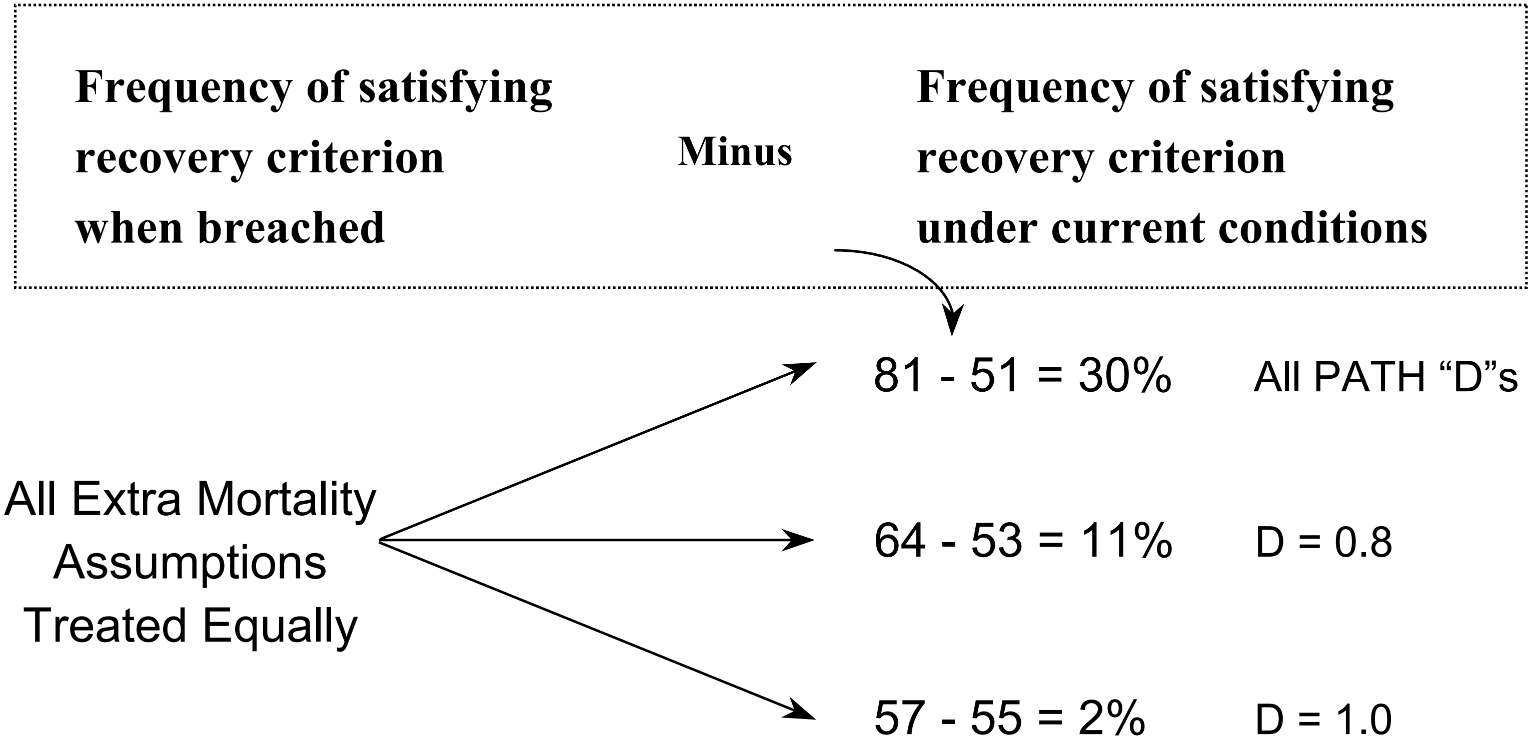
mortality is due to an ocean regime shift, then the gains expected with breaching would be negligible (only 2 percent). NMFS is uncertain about the value of *D*, and only further data can resolve that uncertainty. However, the significance of that uncertainty is unarguable. If D-values are low (as has 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 high (e.g., 0.80 or higher), then the value of breaching depends strongly on what is assumed as the dominant source of extra mortality. Two assumptions are required for breaching to provide only minor benefits relative to transportation: 1) *D* is high (~ 0.8) and 2) nontransported fish do not suffer major extra mortality below Bonneville as a result of the hydropower system.



Note: These data are predicted by the PATH life-cycle model. Solid horizontal line indicates the 48-year recovery criterion.

Figure 4-7. Relationship between Different Combinations of Assumptions and the Average Frequency of Exceeding the 48-Year Recovery Escapement Level

WHAT IS GAINED BY BREACHING?



Frequency of satisfying recovery criterion when breached

Minus

Frequency of satisfying recovery criterion under current conditions

All Extra Mortality Assumptions Treated Equally

81 - 51 = 30% All PATH “D”s

64 - 53 = 11% D = 0.8

57 - 55 = 2% D = 1.0

Figure 4-8. Demonstration of the Increase in the Frequency with Which the 48-Year Recovery Escapement Level is Exceeded Under Breaching (A3) Compared to the Current Condition (A1)

If D = 0.8,

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

1. Assumed Source of Extra	Increased Frequency of Meeting Recovery Goal (under breaching)
1.1 Hydrosystem	19%
1.2 Degraded Stock	6%
1.3 Ocean Regime	2%

Figure 4-9. Sensitivity of the Frequency with Which the 48-Year Recovery Goal is Satisfied (i.e., 48-Year Recovery Escapement Level) under Breaching to Assumptions About the Source of Extra Mortality

5. PATH Analyses of 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 that 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 (the ratio of 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 construction of the Hells Canyon complex of dams 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 before construction of 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., 1998).

5.1 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. (1998).

The Snake River bright (SRB) fall chinook salmon 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). SRB 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. (1998).

5.1.1 Run Reconstructions

Marmorek et al. (1998) 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 counted at the uppermost Snake River Dam (Figure 5-1) 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. Each annual return is made up of contributions from several brood years.

5.2 Adult Harvest and Upstream Passage

5.2.1 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 United States 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 (Chinook Technical Committee, 1988). The approach is based on 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 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 5-1.

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 dominated by the age-2 fish 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 5-2. Estimates indicate that ocean harvest rates have declined from as high as 50 percent in the early 1980s to the current level of roughly 20 to 30 percent.

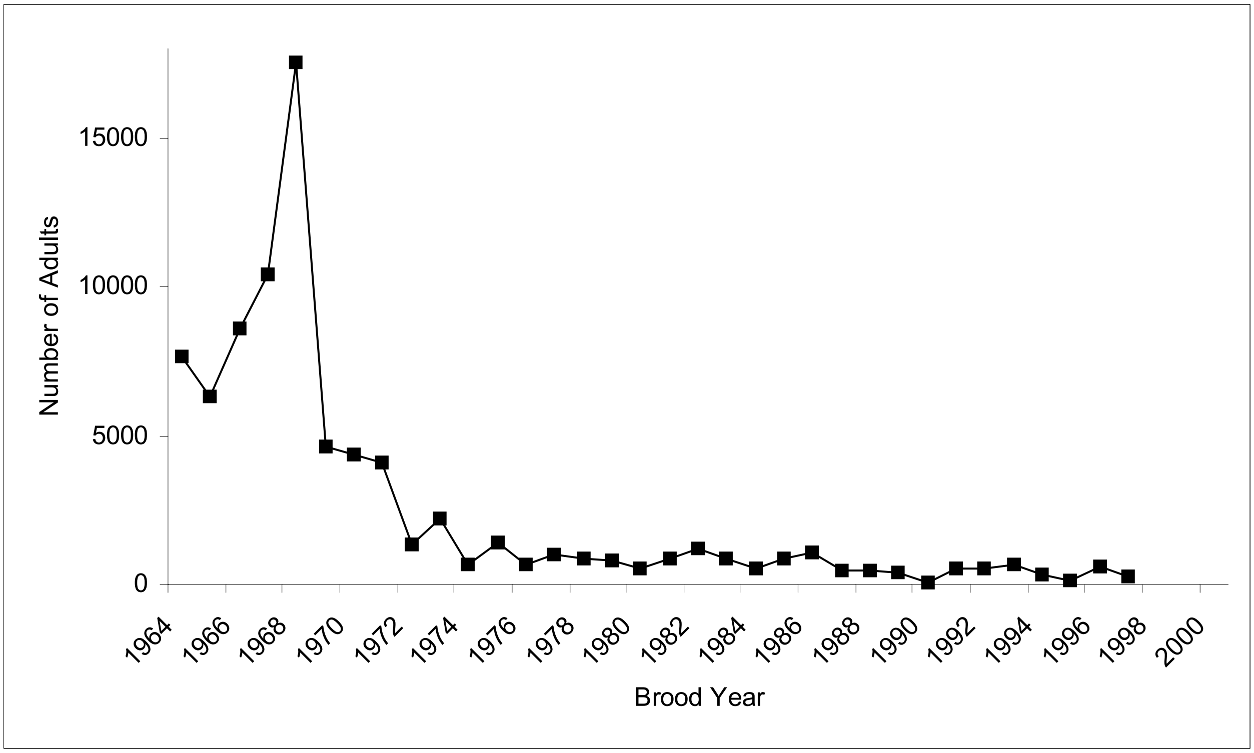


Figure 5-1. Wild Fall Chinook Salmon Spawner Abundance (count at uppermost Snake River Dam) from Run Reconstructions in Peters et al. (1999)

Table 5-1. Availability of 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	--

Note: BY = Broad Year

Table 5-2. 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

Run Year	Subbasin Exploitation Rate		Mainstem (Columbia & Snake Rivers)				Ocean Exploitation Rate (By Age)				
			Conversion Rate		Exploitation Rate						
	Jack	Adult	Jack	Adult	Jack	Adult	2	3	4	5	6
1964	0.000	0.033	1.000	.0380	0.285	0.382					
1965	0.000	0.034	1.000	0.712	0.176	0.519					
1966	0.000	0.039	1.000	0.785	0.076	0.397	0.044				
1967	0.000	0.041	1.000	0.797	0.104	0.499	0.038	0.219			
1968	0.000	0.044	0.658	0.693	0.050	0.358	0.030	0.181	0.447		
1969	0.000	0.051	0.210	0.628	0.065	0.447	0.029	0.141	0.371	0.514	
1970	0.000	0.039	0.262	0.229	0.139	0.472	0.025	0.120	0.210	0.267	0.514
1971	0.000	0.014	0.125	0.206	0.049	0.478	0.025	0.140	0.291	0.345	0.267
1972	0.000	0.096	0.046	0.193	0.056	0.575	0.020	0.136	0.299	0.391	0.345
1973	0.000	0.038	0.080	0.332	0.091	0.530	0.021	0.101	0.279	0.408	0.391
1974	0.000	0.012	0.080	0.107	0.017	0.477	0.014	0.111	0.164	0.205	0.408
1975	0.000	0.006	0.887	0.368	0.134	0.577	0.027	0.100	0.230	0.329	0.205
1976	0.000	0.018	0.649	0.120	0.067	0.489	0.028	0.147	0.160	0.181	0.329
1977	0.000	0.006	0.595	0.395	0.042	0.480	0.019	0.180	0.317	0.360	0.181
1978	0.000	0.000	0.228	0.373	0.034	0.434	0.015	0.073	0.319	0.402	0.360
1979	0.000	0.000	0.370	0.318	0.021	0.415	0.016	0.082	0.151	0.342	0.402
1980	0.000	0.002	0.315	0.290	0.016	0.161	0.014	0.085	0.115	0.107	0.342
1981	0.000	0.008	0.214	0.212	0.010	0.224	0.014	0.059	0.113	0.163	0.107
1982	0.000	0.000	0.347	0.267	0.012	0.139	0.016	0.107	0.085	0.068	0.163
1983	0.000	0.000	0.420	0.407	0.011	0.226	0.023	0.147	0.202	0.215	0.068
1984	0.000	0.000	0.434	0.879	0.024	0.384	0.025	0.147	0.310	0.357	0.215
1985	0.000	0.000	0.734	0.579	0.067	0.397	0.025	0.105	0.223	0.303	0.357
1986	0.000	0.000	0.537	0.379	0.055	0.469	0.015	0.106	0.170	0.169	0.303
1987	0.000	0.000	0.263	0.364	0.037	0.560	0.037	0.156	0.140	0.159	0.169
1988	0.000	0.000	0.738	0.331	0.046	0.524	0.027	0.060	0.288	0.172	0.159
1989	0.000	0.000	0.566	0.372	0.026	0.432	0.038	0.151	0.233	0.227	0.172
1990	0.000	0.000	0.129	0.370	0.028	0.452	0.042	0.059	0.271	0.252	0.227
1991	0.000	0.000	0.691	0.240	0.044	0.276	0.026	0.051	0.138	0.212	0.252
1992	0.000	0.000	0.220	0.503	0.051	0.166	0.020	0.095	0.242	0.204	0.212
1993	0.000	0.000	0.571	0.583	0.050	0.254	0.006	0.079	0.244	0.204	0.204
1994	0.000	0.000	0.879	0.605	0.033	0.155	0.015	0.014	0.229	0.204	0.204
1995	0.000	0.000	0.387	0.323	0.025	0.115	0.016	0.047	0.074	0.169	0.204
1996	0.000	0.000	0.570	0.372	0.039	0.171		0.046	0.000	0.158	0.169
Mean	0.000	0.015	0.491	0.416	0.060	0.383	0.024	0.108	0.218	0.253	0.257
Min	0.000	0.000	0.046	0.107	0.010	0.115	0.006	0.014	0.000	0.068	0.068
Max	0.000	0.096	1.000	0.879	0.285	0.577	0.044	0.219	0.447	0.514	0.514

Note: Ocean exploitation rates were used to expand Columbia River-mouth recruits to account for impacts of ocean harvest.

5.2.2 Upstream Passage

As described in Section 3.2.2, 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 nonharvest losses between Bonneville Dam and McNary Dam are calculated for the aggregate upriver bright run, including the Hanford Reach and Snake River populations (Table 5-2). 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 3.2.2). The problem with the conversion rates in Table 5-2 is that they reflect only 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 wanted 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 a less significant 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.

There are many different methods for estimating upstream survival in fall chinook and these are detailed in Marmorek et al. (1999). The rates vary between 0.48 as the minimum and 0.68 as a maximum (see Table 4.4-13 of Peters et al. [1999]).

5.3 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 starts 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) indicated that the survival rate of fall chinook salmon marked in mainstem spawning and rearing areas approximately 120 kilometers upstream of Lewiston, Idaho, ranged from 40 to 60 percent by the time they had migrated to the 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 millimeters to generally larger than 140 millimeters 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 percent each. 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.

5.4 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 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.

5.4.1 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 historical flow, spill, and elevation data, and the prospective simulations are based on output from the hydroregulation models that describes 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).

5.4.2 Survival to Below Bonneville Dam

5.4.2.1 Reservoir Survival and Influences of Predation

Loss of sub-yearling 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 (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. Following 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 vulnerability (i.e., prey) (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 percent. The survival to Lower Granite Dam of natural fish released near Billy Creek averaged 32 percent. Survival rates for hatchery sub-yearlings 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 are extensive for John Day Reservoir (Poe and Rieman, 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 predator abundance and predator consumption rate parameters in the passage model are 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 Resources 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.

5.4.2.2 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, and 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 Independent Scientific Group (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, 1.0 (the same value used previously in the spring chinook salmon analyses) was adopted as the default value for spill effectiveness at all dams except Dalles Dam.

The mortality of fish that pass a dam via bypass systems was estimated through paired-release experiments at Little Goose Dam that NMFS conducted from 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, personal communication, 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 FGE for the different types of screens used to divert the juveniles from turbines. Two sets of FGEs developed for fall chinook salmon were used in simulations 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. (1998) and in Krasnow (1997).

A portion of the sub-yearling chinook salmon collected in bypass collection facilities at Lower Granite, Little Goose, Lower Monumental, and McNary Dams is 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 before 1982 by NMFS and subsequently by the Corps (Table 5-3). 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 by trucks or barges incur some mortality before release below Bonneville Dam. Studies designed to estimate transport survival for sub-yearling chinook salmon have not been conducted; 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 have to be varied in future simulations to represent uncertainty in direct transportation survival.

5.4.3 Components of Post-Bonneville Dam Mortality

5.4.3.1 Extra Mortality of Transported Fish

In the PATH analyses, specific hypotheses are developed for the relative post-Bonneville survival of transported fish, compared to nontransported fish. Because most Snake River fall chinook juveniles are transported, estimates of the relative post-Bonneville survival of transported and nontransported fish are important in determining the relative efficacy of hydropower actions relying on smolt transportation (i.e. Alternative Actions A2 and A2').

PATH uses the term *D* to denote the ratio of post-Bonneville survival rate of transported fish to that of nontransported fish. A D-value of less than 1.0 suggests that transported fish have lower post-Bonneville survival rates than nontransported fish, while a D-value of greater than 1.0 suggests that transported fish survive better post-Bonneville than nontransported fish. If *D*=1, then both transported and nontransported fish have the same post-Bonneville survival rate.

Table 5-3. Cutoff Dates for Transporting Fall Chinook Salmon Smolts at Lower Granite (LGR), Little Goose (LGS), and McNary (MCN) Facilities

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

For spring/summer chinook, it was possible to use transport:control ratios (TCR) resulting from PIT-tag transportation studies to directly estimate D-values for Snake River fish. However, because no such studies have been done on Snake River fall chinook, an indirect method is necessary to derive a D-value. Transport:control studies have not been possible for fall chinook because there have not been enough returns to estimate survival rates of transported and nontransported fish. The lack of such data to resolve this critical uncertainty points to the need for rigorously designed research, monitoring, and experimental management programs to develop better estimates of the effectiveness of transportation. Because the proportion of fish transported has been consistently high in recent years, experimental manipulation of transportation would provide useful contrast in the data. PATH has begun to define and evaluate such programs, and intends to focus on this in the next year.

PATH subgroups have explored five alternative methods for indirectly calculating a D-value for Snake River fall chinook (sections of this report where methods are described in detail are in brackets). The five alternative methods are listed below, with strengths and weaknesses for each method summarized in Table 5-4:

1. Estimate *D* from TCRs from 1995 PIT-tag data for Snake River fall chinook. SARs are calculated for hatchery smolts PIT-tagged in 1995. The ratio of SAR of transported fish to the SAR of nontransported fish can be used to represent a TCR for that release group.

   TCRs are used to calculate D-values based on the equation:

   $$D = TCR * Vc$$

   where:

   Vc = Survival of inriver migrants from tailrace of collector project to below Bonneville (see note below)

2. Estimate *D* for Hanford Reach fall chinook based on TCRs from transport studies conducted on Hanford fish at McNary Dam from 1978 to 1983 (Section A.2). TCRs are calculated from mark and recovery of freeze-branded smolts, then Ds are calculated using the equation above. Vcs were estimated either from expansion of reach survival estimates or with a passage model (CRiSP).
3. Estimate *D* from Snake River fall chinook spawner-recruit data. *D* was included as a term in the stock-recruit function and a distribution of *D* was estimated based on fits to the historical spawner-recruit data. For prospective simulations, a D-value was selected from this distribution and used in each prospective year.
4. Estimate *D* from Snake River fall chinook spawner-recruit data as above, then adjust based on comparison of spring/summer chinook D-values estimated from spring/summer spawner-recruit data to estimates from spring/summer transport studies. For spring/summer chinook, TCR-based estimates of *D* were generally higher than estimates from spawner-recruit data. These differences were used to inflate the fall chinook Ds estimated from the spawner-recruit data.
5. Estimate a reasonable bound on *D* based on SAR estimates for Snake River fall chinook (primarily Lyons Ferry hatchery fish) and other Columbia River fall chinook stocks.

Summary of D Values

The estimates of *D* that resulted from each of the five methods described above are summarized in Table 5-5. Estimating D-values for Hanford fish from McNary transport data (Method 2) produces qualitatively different estimates than the other three methods. Methods 1, 3, and 4 all produce D-values in the zero to 0.5 range, while method 2 produces D-values ranging from 0.6 to 6.0.

D Hypotheses

The D-values summarized in Table 5-5 were used to develop four alternative D-hypotheses. These hypotheses are intended to reflect hypotheses about the magnitude of *D* in both the retrospective (1965 to 1992) and prospective (1992 to present) periods. The four hypotheses are summarized in Table 5-6, followed by more detailed descriptions and rationales for each.

Table 5-4. Summary of Strengths and Weaknesses of Alternative Methods for Calculating D

Method	Pros	Cons
1	Provides a recent *D* estimate; reflects recent transport conditions Data are specific to Snake River fall chinook Estimated mean *D* consistent with method 4	Only 1 year of data; not representative of all years Anomalous environmental and migration timing conditions in 1995 Some smolts overwinter in hydrosystem Small number of adult returns (adult returns incomplete) = wide confidence limits Estimate is based on hatchery fish
2	Able to estimate *D* directly using transportation studies, similar to spring/summer chinook method Uses multiple years of transportation data D calculated through a T/C has fewer assumptions than *D* calculated through the life-cycle model	Applicability to Snake River fish is limited - spawning, rearing, migration, and transportation conditions/methods different for Hanford fish than Snake River fish Hanford *D* estimated from spawner-recruit data (MLE = 1.0 to 1.14) suggests Hanford fish much more resilient to transportation than Snake R. fish (MLE = 0.02 to 0.05) Results in poorer fit to spawner-recruit data (Section 6.1.2)
3	Uses spawner-recruit data specific to Snake River fall chinook Multiple years of data cover wide range of flow conditions Maximizes historical fit to spawner-recruit data	Prospective D-values based on historical spawner-recruit data, assumes historical transport conditions/methods apply in the future Estimates are influenced by 1990 and 1991 data points (6.1.2, Appendix E) D estimated from spawner-recruit data is negatively correlated with E (spawning effectiveness); adds to uncertainty
4	Uses spawner-recruit data specific to Snake River fall chinook Adjusts for possible bias introduced by estimation method Estimated *D* consistent with method 1	Correction method somewhat arbitrary; difference in spring/summer estimates not necessarily applicable to Snake River fall chinook Estimates are influenced by 1990 and 1991 data points (6.1.2, Appendix E) D estimate from spawner-recruit data is negatively correlated with E (spawning effectiveness); adds to uncertainty
5	Consistent with SARs estimated for Snake R. fall chinook Provides a recent *D* estimate; reflects recent transport conditions Data are specific to Snake River fall chinook Multiple years of data cover wide range of flow conditions	SARs for Snake River fall chinook rely primarily on hatchery fish from Lyons Ferry Hatchery Requires some assumptions to estimate LGR-LGR and BON-BON SARs (don't have good estimates of FGE and survival to Bonneville)

Table 5-5. Summary of T:C and D-values Resulting From Five Different Estimation Methods

Method	T:C Ratio Range (mean)	Vc	D Range (mean)
1a 1995 PIT-tag	0.25 to 2.61 (1.18)	0.20	0.05 to 0.52 (0.24)
1b 1995 PIT-tag	0.74 (1995)	0.20 (1995)	0.15 (1995)
	0.96 (1996)	0.27 (1996)	0.26 (1996)
2 McNary T:C	2.2 to 6.33	0.27 to 0.49	0.6 to 6.0 (1.7)
Est. from S/R data			
CRiSP (Upper)	0.10 to 1.35 (0.15)	0.20	0.02 to 0.27 (0.03)
CRiSP (Lower)	0.10 to 1.35 (0.15)	0.20	0.02 to 0.27 (0.03)
FLUSH (Upper)	0.10 to 1.20 (0.15)	0.20	0.02 to 0.24 (0.03)
FLUSH (Lower)	0.10 to 1.05 (0.25)	0.20	0.02 to 0.21 (0.03)
Adj. Est. from S/R data			
CRiSP (Upper)	0.80	0.20	0.16
CRiSP (Lower)	0.80	0.20	0.16
FLUSH (Upper)	0.60	0.20	0.12
FLUSH (Lower)	0.60	0.20	0.12

Table 5-6. *D* Hypotheses

Scenario	Retrospective *D*	Prospective *D*	Evidence
D1	drawn from posterior distribution of D-values (MLE values around 0.05)	0.24	spawner-recruit data (retrospective), 1995 PIT-tag estimates (prospective)
D2	1.00	1.00	MCN T:C estimates, NMFS analysis of SARs (retrospective and prospective)
D3	drawn from posterior distribution of D-values (MLE values around 0.05)	drawn from posterior distribution of D-values (MLE values around 0.05)	spawner-recruit data (retrospective and prospective)
D4	0.2	0.2	1995 PIT-tag estimates (retrospective and prospective)

Description and Rationale

Hypothesis D1

Several methods were used to estimate *D* for Snake River fall chinook for the retrospective period. The methods all involved indirect estimation procedures and the resultant values were generally low, with means ranging from about 0.04 to 0.24, depending on the method. If this range of values is indeed representative of Snake River fall chinook responses to transportation, then by-and-large that passage strategy was ineffective, or perhaps even detrimental during that era. This is in stark contrast with *D* as estimated for fall chinook passing McNary Dam, which are comprised principally of Hanford and Priest Rapids Hatchery stocks. Estimates for that population averaged near 1.7.

Perhaps the difference in the transport methods employed at Snake River dams and McNary might account for such disparate responses. Over a series of years during the retrospective period, fall chinook smolts were primarily transported by barge from McNary Dam, whereas trucks were the dominant conveyance for Snake River fall chinook. Giorgi (1997) estimated that approximately 15 percent and 85 percent of the sub-yearling chinook were transported via truck at McNary and Lower Granite Dams, respectively. The heavy reliance on trucks at Snake River dams may have been detrimental in two respects. First, trucked fish are not exposed to serial imprinting cues. This may increase the straying rate of the trucked fish upon return and result in low returns to Lower Granite Dam. Presumably inriver migrants adequately imprint and straying is minimized in that segment of the population. These proposed straying dynamics would result in low D-values.

Additionally, the nature of the mark recapture protocols used at McNary and Lower Granite Dams may affect estimates of *D*. The data used to calculate *D* from McNary Dam are CWT recaptures throughout the fisheries and a variety of terminal sampling sites. Any straying effect would not be reflected in the resultant TCR estimates that were employed to calculate *D*. In contrast, the PIT-tag data used to estimate *D* at Lower Granite Dam rely on adults successfully homing to the detector at that site. Increased straying rates that may be associated with trucking would yield low *D* estimates as currently reported for that population.

A second mechanism that could result in poor survival of trucked fish relative to barged counterparts is the nature of the release protocol downstream from Bonneville Dam. From 1977 until 1992, trucked fish were released at the shoreline in the vicinity of either Bradford Island or the Hamilton Island boat ramp. In recent years, concentrations of northern squawfish have been observed in these locations. In an effort to reduce predatory fish consumption of smolts, commencing in the summer of 1993 most trucks containing fish were ferried from the mainland below Bonneville Dam to a mid-channel release site (Corps, 1995). In 1993, some trucks still released fish at Bradford Island (Corps, 1995). By the summer of 1994, all trucked groups were reported as released at a mid-channel site near Dodson, several miles below Bonneville Dam (Corps, 1996). Barged fish were released mid-channel near Skamania Light Buoy, the location of the truck release site.

This hypothesis maintains that prior to 1993 or 1994, transport practices (trucking with shoreline releases) depressed survival of trucked fish and/or exacerbated straying of Snake River fall chinook. This yielded a low value for the *D* estimate on the order of 0.1. This value is consistent with the lower range of estimates produced for the retrospective period. We speculate that the change in release strategy initiated in 1993 increased survival of trucked smolts, resulting in an increase in *D* to 0.24. This estimate is based on the highest mean value as estimated from PIT-tagged fish from the Snake River in 1995. Recent preliminary estimates by NMFS suggest that a higher D-value near 0.8 yields more tractable SAR estimates to Lower Granite Dam. However, if straying associated with trucking remains the primary mechanism depressing *D*, then only the abandonment of this practice will permit further increase in *D*. An exploration of the implications of such a shift to full barge transportation is found in Section 5.3.1.

Hypothesis D2

This hypothesis states that both retrospective and prospective D-values are high (1.0). This follows the precedent set for the spring chinook analysis where lower river stocks were used as surrogates to define the response of the Snake River fish without transportation or passage through the Snake River dams. With a high D-value, the Aikiaki Information Criteria (AIC) scores from the existing life-cycle model will be high, suggesting that under the hypothesis of a high *D*, the trend in extra mortality expressed by a step function and a climate cycle does not capture the underlying trend. In this case, different extra mortality trends and mechanisms need to be explored in the retrospective analysis. The resulting extra mortality, along with a high *D*, would be used in the prospective analysis.

Hypothesis D3

This hypothesis is that the relatively low D-values estimated from the spawner-recruit data in the retrospective period also apply into the prospective period. The hypothesis assumes that possible mechanisms for a low D-value are either related to transportation methods or conditions that will continue into the future, or are related to inherent characteristics of Snake River fall chinook (e.g., small size) that make them less resilient to transportation.

Hypothesis D4

Given the lack of information available to estimate a D-value for Snake River fall chinook, one possible hypothesis is that *D* was 0.2 retrospectively (confidence interval = 0.07 to 0.52). Because there are many factors that can influence transportation effectiveness relative to inriver fish, there is no evidence that the range of D-values will change prospectively. This hypothesis relies on direct estimates of transport:controls from Snake River fall chinook sub-yearlings.

This estimate of *D* is based on PIT-tag recoveries from outmigration years 1995 and 1996 and Vcs from the FLUSH passage model. The recoveries from the two years include 44 adult returns for 1995 (ages 2 to 4) and 30 adult returns for 1996 (ages 2 to 3). Transport and control SARs were generated for all releases of sub-yearlings above Lower Granite Dam for the entire outmigration season (See section A.1.2). Estimated TCRs for 1995 and 1996 were 0.74 and 0.99, respectively. FLUSH Vcs were 0.197 and 0.269 for 1995 and 1996, respectively. Although the recoveries included detections from the group released at Lower Granite Dam but detected at Lower Granite, Little Goose, Lower Monumental, and McNary dams, only Vcs from Lower Granite Dam were used. This assumption likely results in a slight under-estimation of Vc, which could cause a similar under-estimation of *D*. However, because 70 to 80 percent of the detections were at Lower Granite and Little Goose, this assumption likely has little effect on our estimates of *D* (the wide confidence interval should capture this potential bias).

The variance for the TCR was calculated as:

$$\mathrm{Var}(\ln[T/C]) = 1/n_t + 1/n_c - 1/N_t - 1/N_c$$

Where:

n_t = number of transport juvenile releases
n_c = number of control juvenile releases
N_t = number of transport adult returns
N_c = number of control adult returns

The confidence interval was estimated from 2* S.E. (S.E. ≈ Std Dev.) of the TCR. The point estimate and the confidence interval for transport:control were used to estimate *D* for 1995 and 1996 using FLUSH estimates of Vc for 1995 and 1996.

The 1995 fall chinook sub-yearling releases were from fish collected at Lyons Ferry hatchery, reared at Klickitat hatchery, and then trucked and released above Lower Granite Dam. Although this treatment may affect the overall SAR, the transport and control fish had the same treatment. Therefore, the transport:control should be a reasonable approximation for Snake River fall chinook sub-yearlings. The 1996 fall chinook sub-yearling releases were from Lyons Ferry hatchery-reared fish.

When SARs are estimated on a brood year basis with complete age structure applied to recruits (in contrast to assuming that all recruits are 4-year-olds), they appear to show noticeable increases starting in the 1991 outmigration year. This does not correspond to the hypothesis that SARs increased in 1993 and 1994 as a result of the onset of offshore releases of transported fish (as implied in *D* hypothesis #1). Further, the transport:control estimates from 1997 sub-yearling releases of Lyons Ferry Hatchery and 1998 jack returns (T. Cooney, NMFS, 3/9/99 memo to files) appear to be generally consistent with the 1995 and 1996 transport:control data. The TCR for the 1997 outmigration, using estimates of nondetected smolts as controls, was 0.65.

Implementation of D-Hypotheses

In the current round of modeling, D-values were implemented as fixed values with no uncertainty, except for hypothesis D3 (i.e., 0.24 for D1, 1.0 for D2, and 0.20 for D4; under hypothesis D3 D-values are drawn from the posterior distribution of D-values estimated in the life-cycle model). However, given the amount of uncertainty inherent in these estimates (which stems from the lack of transport studies for fall chinook), a better approach would be to include some variability in these D-values rather than assuming that they are a constant. Future analyses could include such variability, or at least conduct sensitivity tests to see the effect on results of including variability.

One approach to do this would be to specify a distribution of D-values to apply prospectively, then draw from that distribution in each year. Hypothesis D4 has specified such a distribution; similar methods for calculating confidence intervals could be used with hypotheses D1 and D2 to derive similar distributions of D-estimates.

5.4.3.2 Extra Mortality of Nontransported Fish

Extra mortality is mortality that is not captured by the passage model and assumptions about the effectiveness of transportation. Extra mortality may or may not exist depending on the life-cycle model employed. If the recent declines in productivity are hypothesized to be accounted for by increases in passage mortality and poor effectiveness of transportation, then there is not extra mortality. Extra mortality in the fall chinook life-cycle model is modeled using the *STEP* term, which represents the 1975 brood year climate regime shift (see Annex A, page A-7) and is assumed to be zero prior to 1970 or 1976, depending on which hypothesis is employed. Afterwards, it is either assumed to take on the value of zero (fall-D model), or it takes the value of the estimated change in productivity not accounted for by the passage mortality or transportation effectiveness (fall-S model). The fall-D model, for which the transportation effectiveness (D) is estimated from the spawner-recruit numbers, *STEP* is assumed to be zero. With the fall-S model, the *STEP* factor is estimated from retrospective data. In prospective simulations with the fall-S model, there are three alternative hypotheses about future values of *STEP*. These three hypotheses are analogous to the three extra mortality hypotheses defined for spring/summer chinook. Detailed descriptions, rationales, and evidence for these hypotheses are provided in Section A.3.3 of the *Preliminary Decision Analysis Report for Spring/Summer Chinook* (Peters et al., 1999), and in Section 4.2.3 of the *PATH Weight of Evidence Report* (Marmorek and Peters, 1998b).

Regime Shift Hypothesis

Extra mortality is an interaction with a long-term oscillation in climate that shows a climate regime shift approximately every 30 years. In this century, the regime shifts (or polarity switches) occurred in 1925 (to warm/dry); 1947 (to cold/wet); and 1977 (to warm/dry). The signatures of a recurring pattern of interdecadal climate variability are widespread and detectable in a variety of Pacific basin climate and ecological systems. These climate oscillations affect ocean temperatures and currents, which affect distributions of predators and prey, and broad-scale weather patterns over land masses, which affect temperatures, rainfall, snowpacks, and flows. The regime shifts show an inverse pattern in salmon production between the Alaskan stocks and West Coast stocks over the 20th century (Hare et al., 1999). While Alaskan stocks showed a dramatic increase corresponding to the 1977 regime shift, many West Coast stocks showed declines.

Modeling the future climate is difficult because it is uncertain when the next regime shift will occur. However, over the last century, a 60-year cycle fits the average climate oscillation fairly well. Therefore, in prospective simulations, *STEP* oscillates in a 60-year cycle between the values of 0.0 (good climatic periods) and a value selected from the posterior distribution for *STEP* (poor climatic periods). The cycle turned non-zero in brood year 1976 (ocean year 1977).

Hydro-Related Hypothesis

STEP will continue in the future at a value selected from its posterior distribution, assuming a change in brood year 1976 (or alternatively, 1970), unless the Snake River dams are removed, in which case *STEP* will equal 0.0. This is analogous to the method for spring/summer chinook, described in Appendix H of the *PATH Weight of Evidence Report* (Marmorek and Peters, 1998b), which resolves some of the problems with making post-Bonneville survival proportional to inriver survival (see Section 4.2.3 of *PATH Weight of Evidence Report*). The hypothesis is that the extra mortality was caused by the Snake River dams. With this hypothesis, drawdown of John Day Dam alone would not change extra mortality.

"Here to Stay" Hypothesis

STEP will continue in the future at a value selected from its posterior distribution, again assuming a change in brood year 1976.

5.4.4 Inriver and Ocean Harvest

The fall chinook harvest workgroup developed six different scenarios for future ocean and inriver harvest. The harvest workgroup included Phaedra Budy (USFWS), Howard Schaller (Oregon Department of Fish and Wildlife [ODFW]), Olaf Langness (Washington Department of Fish and Wildlife [WDFW]), Tom Cooney (NMFS), Jim Norris (University of Washington—Bonneville Power Association), and Mike Matelywich (Columbia River Intertribal Fish Commission).

Ocean harvest scenarios were coupled with either the existing inriver harvest schedule or a conservation cutoff-based inriver harvest schedule and are shown in Table 5-7.

Table 5-7. Ocean and Inriver Harvest Scenarios

Scenarios		Ocean	Inriver
HARV1	Baseline	Sample from ocean exploitation rates -return years 1985-1996	Existing inriver harvest schedule
HARV2a	15% ocean increase	Increase ocean exploitation rates by 15% -return years 1985-1996, sample	Existing inriver harvest schedule
HARV2b	15% ocean decrease	Decrease ocean exploitation rates -return years 1985-1996 by 15%, sample	Existing inriver harvest schedule
HARV3	50% ocean reduction	Reduce ocean exploitation rates by 50%-return years 1985-1993, sample	Existing inriver harvest schedule
HARV4	50% ocean reduction 50% inriver reduction and conservation cutoff	Reduce ocean exploitation rates by 50%-return years 1985-1993, sample	50% reduction inriver harvest rates for lower tiers, upper tier harvest rates do not occur until recovery goal is exceeded by 50%
HARV5	75% ocean reduction 50% inriver reduction and conservation cutoff	Reduce ocean exploitation rates by 75%-return years 1985-1993, sample	50% reduction inriver harvest rates for lower tiers, upper tier harvest rates do not occur until recovery goal is exceeded by 50%

HARV1: The HARV1 scenario is the base case where future ocean harvest rates are sampled from the historical ocean exploitation rates for return years 1985 through 1990. These years were chosen because they reflect the implementation of the United States vs. Canada Pacific Salmon Commission (PSC) treaty in 1985 and include a range of exploitation levels that likely bracket the range we might expect to see in the future.

Under HARV1, PSC treaty ocean exploitation rates are coupled with a schedule meant to reflect the existing inriver harvest schedule. The inriver harvest schedule is based on the 1996 to 1998 Inriver Harvest Agreement (United States vs. Oregon) guidelines. Harvest rates are determined by both the Snake River bright (SRB) run size and the healthy upriver bright (URB) run size, since both enter the river at the same time and are harvested primarily in the same fisheries. The URB stock is modeled simultaneously with the SRB stock for the purpose of determining SRB harvest rates. Under drawdown actions (A3, B1), upstream conversion rates affect the harvest rate, as fewer fish are required to meet Lower Granite recovery standards when upstream survival increases. Thus, under drawdown actions, the inriver schedule is adjusted for increased upstream conversion rates via the ranges of recruits in each harvest tier.

HARV2a -b: These scenarios equate to a 15 percent increase and decrease, respectively, in ocean exploitation rates, where the change is applied uniformly to all age classes. Under this scenario, ocean exploitation rates are sampled from the same return years described above (1985 through 1996).

These scenarios were developed during the first-stage fall chinook analyses and loosely correspond to the range of harvest rates reported in the February 10, 1998, draft United States proposal to the PSC for managing major bilateral ocean fisheries. However, because the management proposal is based on legal catch and 15 percent was applied to all age classes, this scenario likely overestimates the effect of this range of harvest. Further, because PSC management agreements are currently under negotiation and have been for several years, it is impossible to predict the actual management scenario that will be used in either the near or distant future.

The HARV 2a-b ocean harvest scenarios were coupled with the existing inriver harvest schedule described above.

HARV 3, 4, 5: These scenarios were developed during the second stage of fall chinook analysis and were not meant to reflect any specific management action. Instead, the reductions were included to represent any dramatic reduction in ocean harvest rates compared to rates observed since the initiation of the PSC treaty. HARV 3 and 4 equate to a 50 percent reduction in the average brood exploitation rates for brood years 1981 to 1989, and HARV 5 is a 75 percent reduction in that rate (approximate return years 1985 to 1993). These reductions might be possible, for example, if one of the major parties (US South, US North, or Canada) were to eliminate a large PSC fishery that impacts Columbia River bright chinook, if a selective fishery were implemented coast wide, or with some combination of both reductions and selective fisheries. Brood years 1981 (approximately 1985 return year) through 1989 were chosen for these dramatic reductions because they reflect the time period after the PSC treaty was initiated but before Canada started substantially reducing its ocean fisheries off West Coast Vancouver Island and elsewhere.

Reductions were applied across the four age classes and brood years with an age specific reduction factor. The brood year exploitation rates were first reduced by 50 percent and 75 percent, and then the average brood year exploitation rate was calculated. The proportion of mortality at age, on average, was also calculated for the baseline data and under the reduced scenarios, and a set of reduction factors was estimated for application to the age-specific ocean exploitation rates. These reduction factors provided the desired (50 percent and 75 percent reduced) average-brood ocean exploitation rate and retained the distribution of mortality (minimized the sum of squares) across ages for the SRB and Hanford/Yakima Upriver Brights (HYURB) stocks.

The HARV3 50 percent ocean reduction scenario was coupled with the existing inriver schedule described above.

HARV4 and 5 ocean scenarios were coupled with a conservation cutoff-based inriver schedule. This schedule includes a dramatic reduction in harvest rates at low SRB run sizes. For the conservation-based schedule, harvest rates in the lower tiers (lower ranges of SRB return size) are restrained to 50 percent of the existing harvest levels. Harvest rates are not allowed to increase as a function of SRB run size until the recovery goal at Lower Granite Dam can be met. These conservation-based harvest levels in the lower tiers are slightly less than ceremonial/subsistence harvest levels. As described above for the existing harvest schedule, under drawdown actions, the inriver schedule is adjusted for higher upstream conversion rates through the range of recruits in each harvest tier.

5.5 Analysis of Hydrosystem Management Alternatives

The assessment of the potential impacts of alternative management actions involving the lower Snake River 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 River dams were constructed. Alternative assumptions regarding biological mechanisms, climate/environmental effects, and the effects of year-by-year actions were 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. 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 were 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.

5.5.1 PATH Results Regarding Management Actions

The fall chinook salmon PATH 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. PATH analyses have considered several hydrosystem management alternatives, including transportation actions and drawdown actions. PATH model response variables for these different actions include trends in projected numbers of spawners over time, and average survival and recovery frequencies over short- and long-time scales. Thus far, the prospective modeling indicates that for all of the management actions analyzed, the fraction of model runs meeting or exceeding the short- and long-term survival and recovery escapement levels is generally high (Table 5-8). Scenarios that failed to meet the recovery standards included transportation actions that assumed relative survival of transported fish does not change in the future. The projected number of spawners over a 100-year simulation was quite variable among the various scenarios. Average escapements were greater for drawdown actions than for transportation actions (Table 5-8). These results varied depending on assumptions regarding survival of transported fish. When survival of transported fish was assumed to be low, average escapement levels for the transportation actions were also low, but were high for drawdown actions. When survival of transported fish was assumed to be high, escapement levels for all actions were intermediate.

PATH also conducted preliminary analyses for management actions that alter habitat and harvest. Sensitivity analyses suggest the median number of spawners would increase by 40 to 50 percent with an increase in habitat. This benefit arises because fall chinook are mainstem spawners, and breaching would open up spawning habitat as the reservoirs were drained. Marmorek et al. (1998) estimate a 77 percent increase in habitat carrying capacity for fall chinook as a result of breaching. However, this 77 percent 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).

5.5.2 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 (ESU) (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. Under the best of scenarios, drawdown of dams in the lower Snake River (A3) could not recover even one-quarter of that original amount.

Nonetheless, drawdown of the lower Snake River facilities 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 areas 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. A model is not necessary to conclude that an increase in spawning habitat on the order of 70 to 80 percent could markedly enhance fall chinook salmon survival and recovery prospects. 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.

Table 5-8. Summary of Major Quantitative Results for Alternative Hydrosystem Actions

Performance Measure	Actions	D Hypotheses (retrospective/prospective D-value)			
		D1 (0.05/0.24)	**D2 (1.0/1.0)**	**D3 (0.05/0.05)**	**D4 (0.20/0.20)**
Number of runs per action / *D* hypothesis[1/]	A1	2	6	2	6
	A2’	2	6	2	6
	A3	16	48	16	48
	B1	32	96	32	96
Average spawning escapement over 100-year simulation period	A2	5,028	5,259	2,131	2,328
	A2’	5,515	6,273	2,151	2,535
	A3	21,312	8,325	20,842	15,425
	B1	24,055	9,961	23,553	17,695
Probability of exceeding survival escapement threshold, 24 years	A2	0.99	0.94	0.80	0.90
	A2’	0.99	0.95	0.73	0.89
	A3	0.99	0.94	0.89	0.92
	B1	0.99	0.94	0.89	0.92
Probability of exceeding survival escapement threshold, 100 years	A2	1.0	0.96	0.80	0.92
	A2’	1.0	0.98	0.72	0.93
	A3	1.0	0.97	0.97	0.98
	B1	1.0	0.98	0.97	0.98
Probability of exceeding recovery escapement threshold, 24 years	A2	0.86	0.70	0.26	0.34
	A2’	0.90	0.78	0.27	0.38
	A3	1.0	0.84	1.0	1.0
	B1	1.0	0.86	1.0	1.0
Probability of exceeding recovery escapement threshold, 48 years	A2	0.87	0.68	0.28	0.34
	A2’	0.93	0.77	0.30	0.40
	A3	1.0	0.83	1.0	1.0
	B1	1.0	0.88	1.0	1.0
Fraction of runs exceeding survival and recovery standards	A2	2/2	6/6	0/2	1/6
	A2’	2/2	6/6	0/2	1/6
	A3	16/16	41/48	16/16	48/48
	B1	32/32	85/96	32/32	96/96

1/ More runs are required for drawdown actions because of the uncertain factors that are specific to drawdown (e.g., length of transition period, survival rate in near-natural river).

6. PATH Analyses of Steelhead

Information on Snake River steelhead (*O. mykiss*) is sketchy because it is difficult 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, the limited quantitative data for steelhead can be supplemented with the spring/summer chinook salmon PATH analyses and inferences. There are, of course, extrapolation limitations 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 fresh water 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 observation of a bimodal migration of adult steelhead at Bonneville Dam and differences in age (1-ocean versus 2-ocean) and adult size among Snake River steelhead. Adult A-run steelhead enter fresh water from June to August; as defined, the A-run passes Bonneville Dam before 25 August (Columbia Basin Fish and Wildlife Authority [CBFWA], 1990; Idaho Department of Fish and Game [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 predominantly 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 to 10 centimeters 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; Columbia River Fish Mitigation Program Technical Advisory Committee [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; 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 juveniles typically emerge from the gravel 2 to 3 weeks after hatching (Barnhart, 1986).

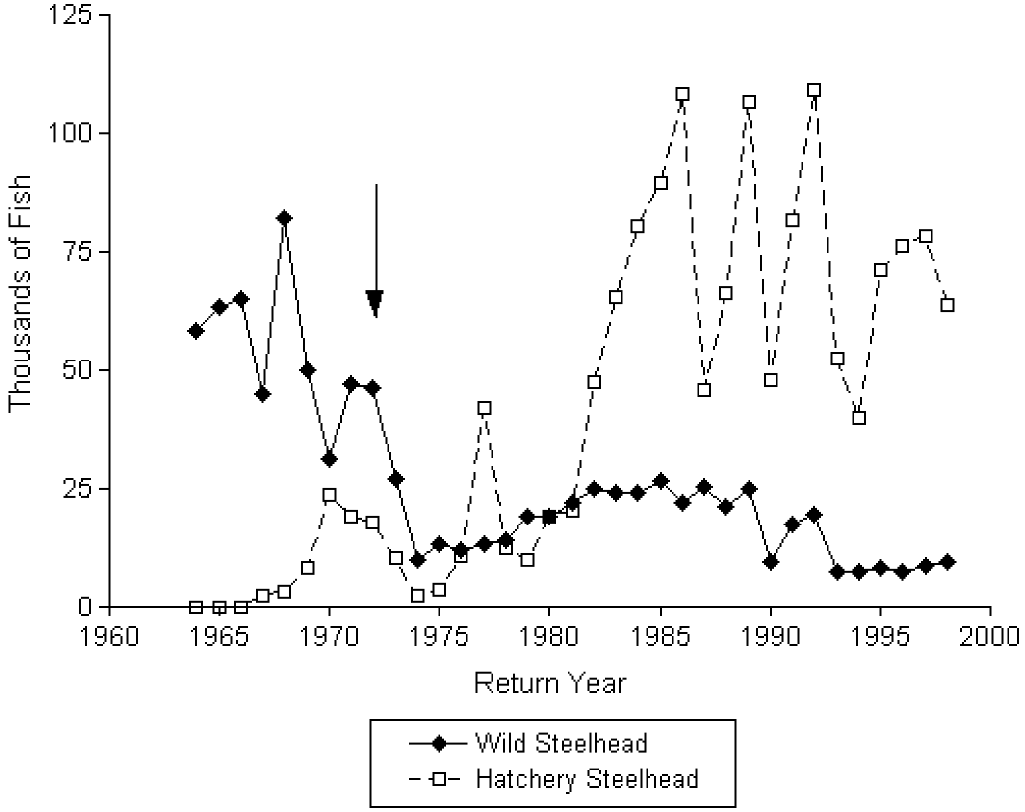
Unlike Pacific salmon, steelhead can spawn multiple times before death. However, it is rare for steelhead to spawn more than twice before dying; most that do so are females (Nickelson et al., 1992). Prior to construction of most lower Columbia River and lower Snake River dams, the proportion of repeat-spawning summer steelhead in the Snake and Columbia rivers was less than 5 percent (3.4 percent [Long and Griffin, 1937]; 1.6 percent [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) is 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 centimeters total length and migrate to the ocean in the spring (Meehan and Bjornn, 1991).

The Snake River Evolutionarily Significant Unit generally matures after 1 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).

6.1 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 6-1). 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 4-1). 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.



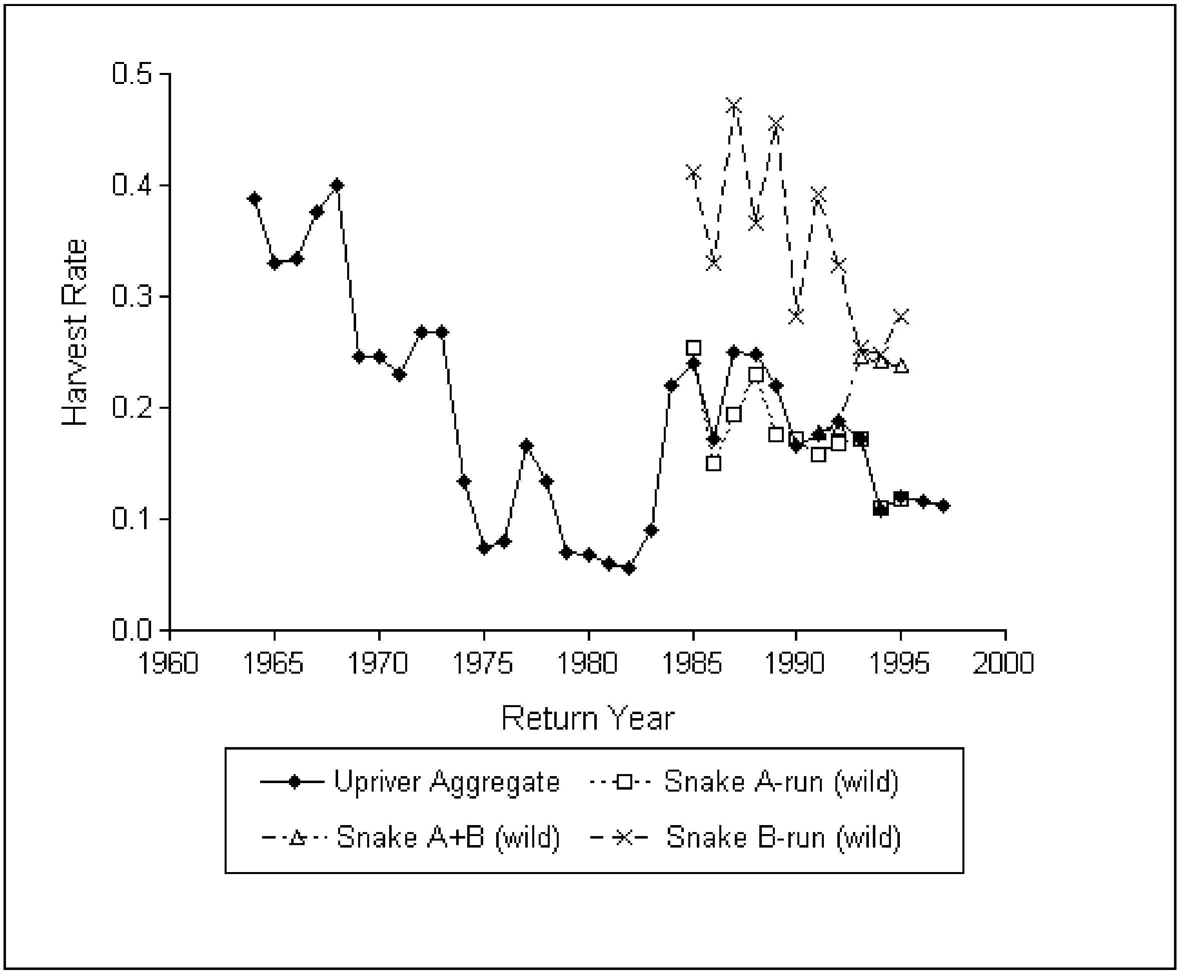
Note: Uppermost dams were Ice Harbor from 1964 through 1968; Lower Monumental during 1969; Little Goose from 1970 through 1974; and Lower Granite in all subsequent years. Arrow represents construction of Dworshak Dam, which blocked access to the North Fork Clearwater River, a significant B-run steelhead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

Figure 6-1. Estimated Returns of Adult Wild and Hatchery Steelhead to the Uppermost Dam on the Lower Snake River

6.2 Adult Harvest and Upstream Passage

6.2.1 Harvest Rates

Snake River steelhead are not targeted by ocean fisheries, and ocean harvest of steelhead is effectively nonexistent. Columbia River harvest rates have varied as a function of run size (Figure 6-2). 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 percent (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 percent. From 1984 through 1993, aggregate harvest rates increased to 16 to 25 percent, and then dropped again to 10 to 11 percent after 1994. This description of aggregate harvest rates is representative of mainstem harvest of wild A-run steelhead but underestimates the



Note: 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; Marmorek et al., 1998). For comparison with other figures, "return year" is, on average, "migration year" + 1 for Snake River steelhead.

Figure 6-2. Harvest Rates for Columbia River Basin Steelhead

wild B-run mainstem harvest rates, which have ranged from approximately 25 to 47 percent since the mid-1980s (TAC, 1997).

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 percent, whereas the wild Snake River steelhead harvest rate has averaged 21.6 percent (Marmorek et al., 1998).

6.2.2 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 turnoffs 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 percent (Ross, 1998; Marmorek et al., 1998). This is similar to the estimate of approximately 76 percent for spring/summer chinook salmon from the same studies. Translated into a mortality rate, it represents approximately 3 percent mortality per hydropower facility.

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-facility survival of 97 percent (= $0.79^{1/8}$) occurred before 1968, increasing the number of dams from four to eight would have decreased passage survival about 10 percent, from 89 percent (= 0.97^{4}) to 79 percent. However, the greatest decline in spawner returns occurred between 1972 and 1974 (Figure 6-1), 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 during that period (from between 23 and 40 percent to between 6 and 13 percent), 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.

6.3 Egg-to-Smolt Life Stage

The egg-to-outmigrating smolt stage for Snake River steelhead covers at least 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 useful information exists 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 pre-spawning 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 River 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.

6.4 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., 1998) (Figures 6-3 and 6-4). 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.

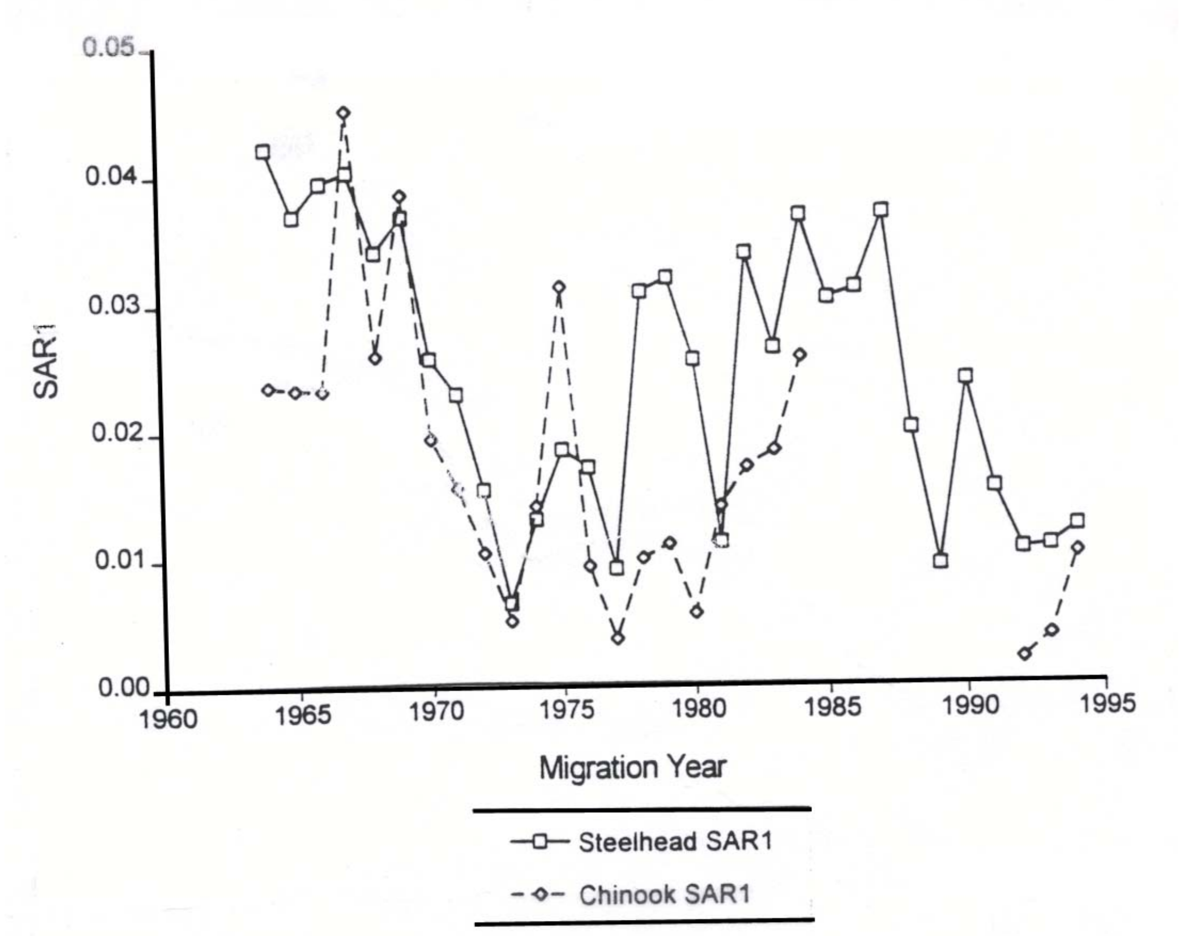
6.4.1 Direct Survival to Below Bonneville Dam

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 uppermost Snake River facility (Lower Granite) to below Bonneville Dam 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, making relatively simple assumptions, by expanding average per-facility survival to reaches that were not included in the study (Smith and Williams, 1999) (Figure 6-5). 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 percent in the case of spring/summer chinook salmon—for which data exist to quantify the over-estimation of bias [Marmorek et al., 1998]). Figure 6-5 has been adjusted for this effect.

The pattern of downstream migrant survival estimates displayed in Figure 6-5 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 four to six dams during the late 1960s. Because a large proportion of steelhead has been transported since the late 1970s, the total direct survival of combined transported and inriver migrants has been even higher than that indicated in Figure 6-5 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



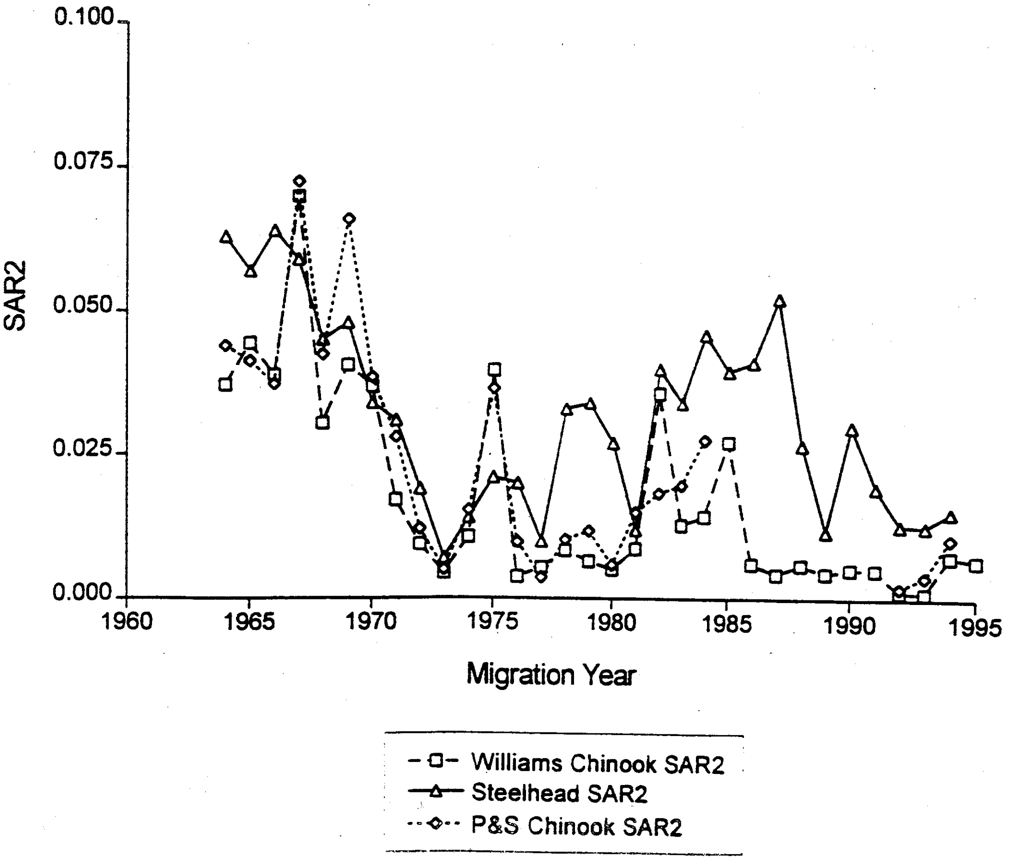
Note: Spring/summer chinook salmon escapement SAR (Chinook SAR1) is displayed for comparison. Estimates from Petrosky (1998) and Petrosky and Schaller (1998).

Figure 6-3. Estimates of Escapement SAR (to upper dam) for Snake River Steelhead (Steelhead SAR1)

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 Dam 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.

6.4.2 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



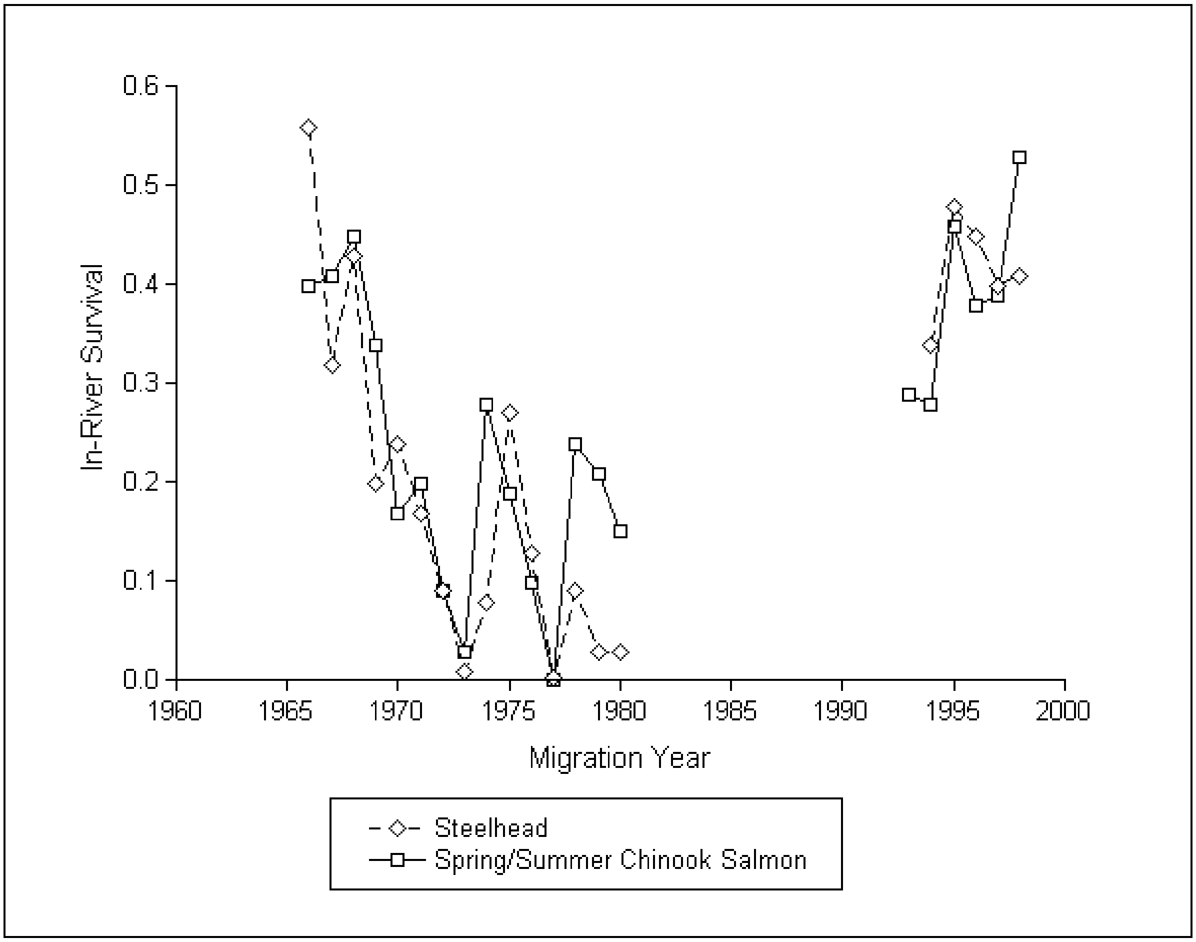
Note: Two estimates of spring/summer chinook salmon Escapement + Harvest SAR are displayed for comparison. Estimates for steelhead SARs are from Petrosky (1998); estimates for chinook salmon SARs from Petrosky and Schaller (1998) and Williams et al. (1998b).

Figure 6-4. Estimates of Escapement + Harvest SAR (Which Counts Harvested Fish Towards Escapement) for Snake River Steelhead (Steelhead SAR2)

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.

6.4.2.1 Climate Effects As a Factor in Survival Below Bonneville Dam

Coronado-Hernandez (1995) noted strong covariation in survival rate and SAR, as inferred from CWT returns among 67 steelhead hatchery stocks distributed throughout the Pacific Northwest (Oregon, Washington, Idaho, and British Columbia). An increase in survival from the mid-1970 through mid-1980 brood years and a decline beginning with late-1980 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 outmigration 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



Note: These estimates are 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 inriver passage experience of PIT-tagged fish compared to the experience of fish in the run-at-large (see text). Note that the inriver 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 6-5. NMFS Reach Survival Estimates

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. (2000) 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 before 1990 were associated with the 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) 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 through 1977 (Figure 2-4, 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.

6.4.2.2 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 Dam. 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 nontransported fish
- general delayed impacts on both transported and nontransported 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 4.4 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 Nontransported Fish

Sandford and Smith (in press) 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 Dam mortality is not equivalent for all fish migrating inriver 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 nontransported fish, as a result of hydrosystem passage, have been proposed and are described in Marmorek et al. (1998).

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

As was the case with spring/summer chinook salmon, several alternative hypotheses explain the extra mortality in 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 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 6-1). 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., a lag in genetic consequences or in ecological interactions may occur that are mediated through changes in habitat quality).

6.5 Examining Alternative Management Actions

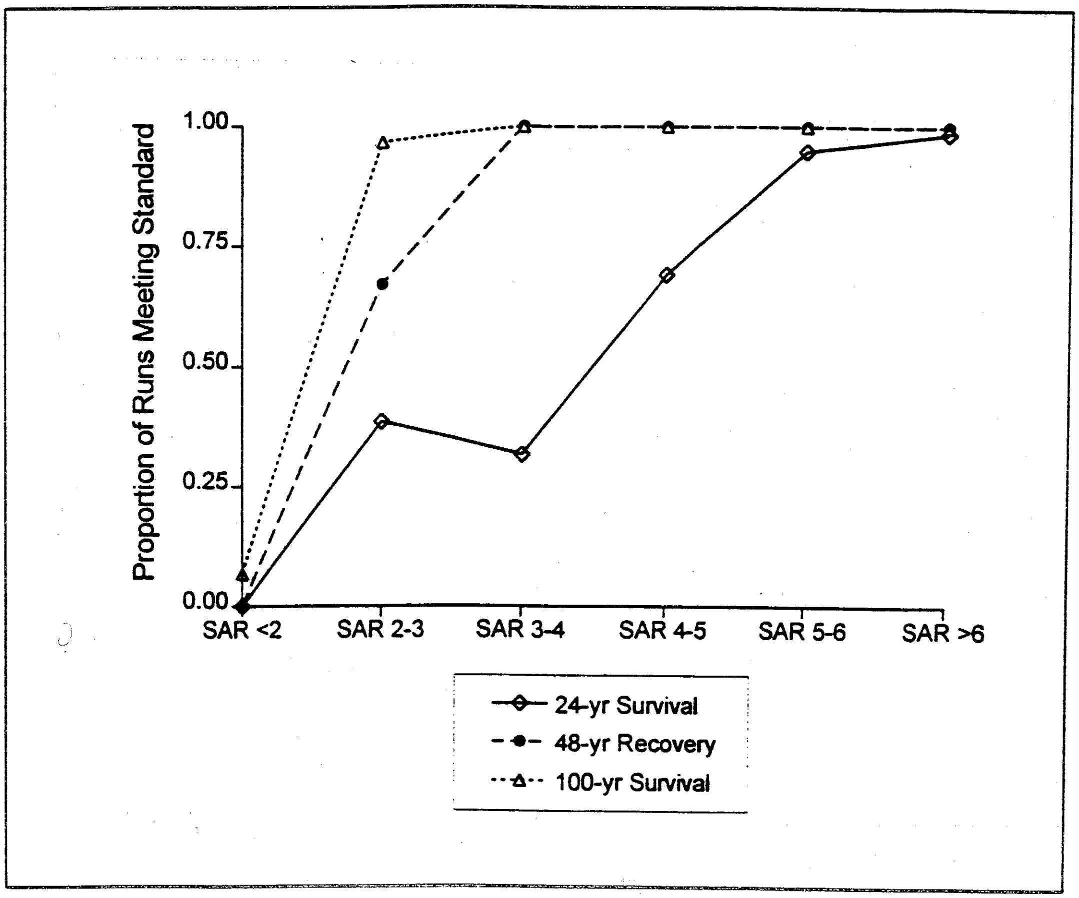
The potential effects on steelhead of implementing alternative actions to address Snake River hydrosystem impacts were not analyzed through the PATH process in the same manner as the effects on spring/summer chinook salmon. Rather, conclusions regarding steelhead were derived by inference from the spring/summer 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 SARs. Assume that, if this correspondence exists for Snake River spring/summer chinook salmon, it will also exist for Snake River steelhead.
2. Define a historical range of Snake River steelhead SARs 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 if:
    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, then historical SARs are a reasonable proxy for an acceptable probability of survival and recovery in Snake River spring/summer chinook salmon and this approach extends to Snake River steelhead.

8. Assume further that if:
    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 hydrosystem survival;
    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) and 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.

Adopting the logic embodied in the seven-step process discussed above in conjunction with what is known directly about steelhead, some conclusions can be drawn. 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 6-6). To ensure that populations remain above survival thresholds over the next 24 years, escapement SARs that are somewhat higher than those observed during the historical period may be required.
2. NMFS agrees with PATH that, based on the information presented in tables 6-1 and 6-2, 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 judgment 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., 1998), 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. Importantly, PATH has not quantitatively considered the effects of



Note: For example, for model runs resulting in a simulated median escapement SAR between 3.0 and 3.99, slightly more than 30 percent of these runs meet the 24-year survival criterion, slightly less than 70 percent 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 percent, certainty of meeting the 48-year recovery criterion requires a median escapement SAR of at least 4 percent, and certainty of meeting the 24-year survival criterion requires a median escapement SAR greater than 6 percent.

Figure 6-6. 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

reduced harvest rates on steelhead, which is a plausible management action that could contribute substantially to steelhead recovery (see Section 8).

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, it is possible that actions which would fail to meet survival and recovery criteria for spring/summer chinook salmon would succeed for steelhead.

Table 6-1. Smolt-to-Adult Return Rate (SAR) Estimates to Upper Dam (Escapement SAR)

	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

Sources: Petrosky, 1998; Petrosky and Schaller, 1998.

Note: These estimates represent historical and recent periods for Snake River spring/summer chinook salmon and Snake River steelhead.

Table 6-2. Smolt-to-Adult Return Rate (SAR) Estimates to Upper Dam, Adjusted for Harvest (Escapement + Harvest SAR)

	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

Sources: Petrosky, 1998; Petrosky and Schaller, 1998.

Note: These estimates represent historical and recent periods for Snake River spring/summer chinook salmon and Snake River steelhead.

7. PATH Analyses of 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 effect 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 SAR life stages.

7.1 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 ESU. 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 ESU.

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 7.2.2.3. 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 ESA (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 7-1; ODFW and WDFW, 1998). The IDFG counted adults at a weir in Redfish Lake Creek from 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, personal communication, Fishery Biologist, IDFG, December 1, 1998).

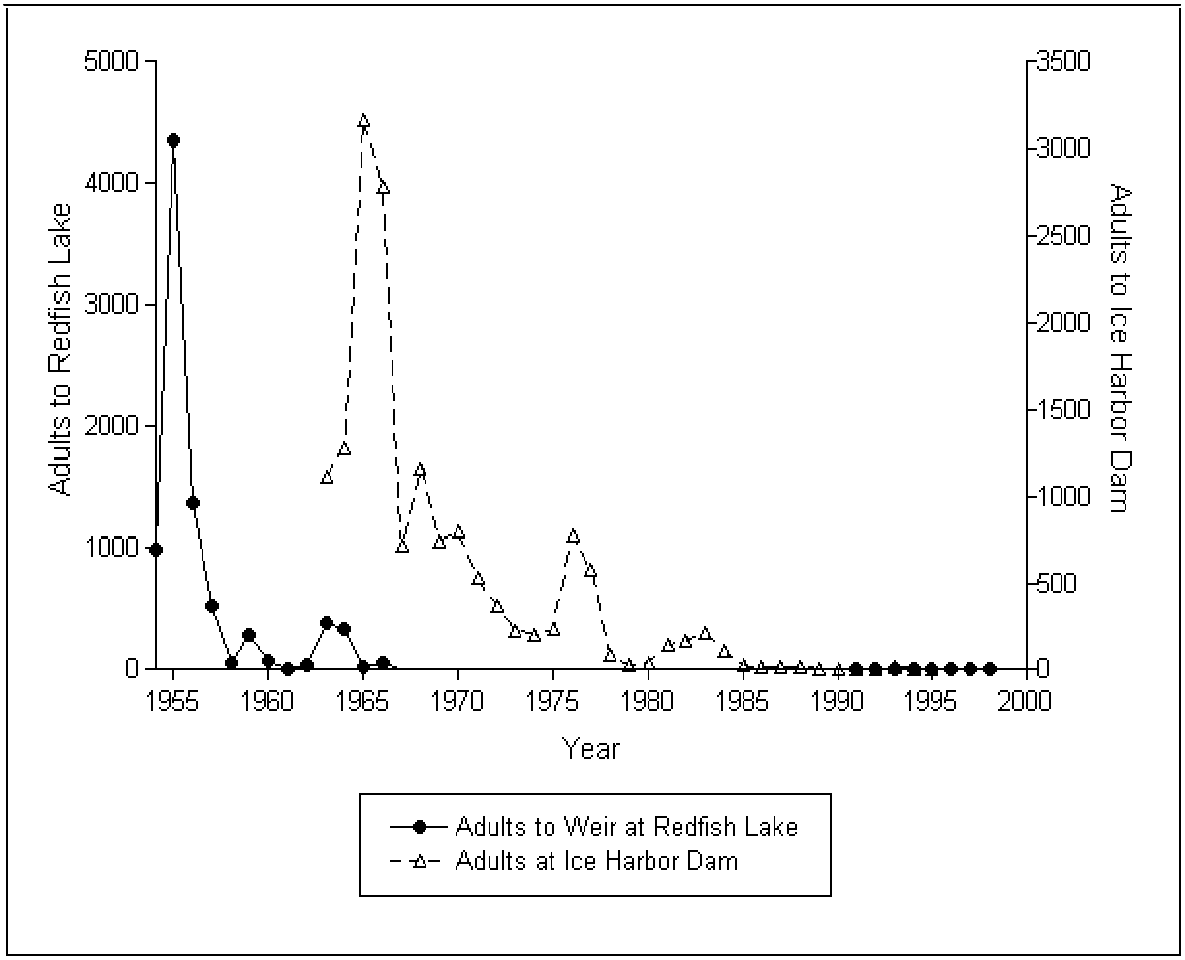
7.2 Adult Harvest and Upstream Passage

7.2.1 Harvest

Although historical mainstem harvest rates for Snake River sockeye salmon have been variable, they were generally higher before, rather than after, the completion of the hydrosystem (Figure 7-2). Annual mainstem harvest averaged 40 percent of adults that returned to the Columbia River mouth (range = zero to 86 percent) before 1974 and 9 percent (range = zero to 49 percent) 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 increased. 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, 1998). Sockeye salmon fisheries are now managed according to the 1996 to 1998 Management Agreement, which allows impacts on sockeye salmon of no more than 1 percent in the non-Indian commercial and recreational fisheries combined.

7.2.2 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).



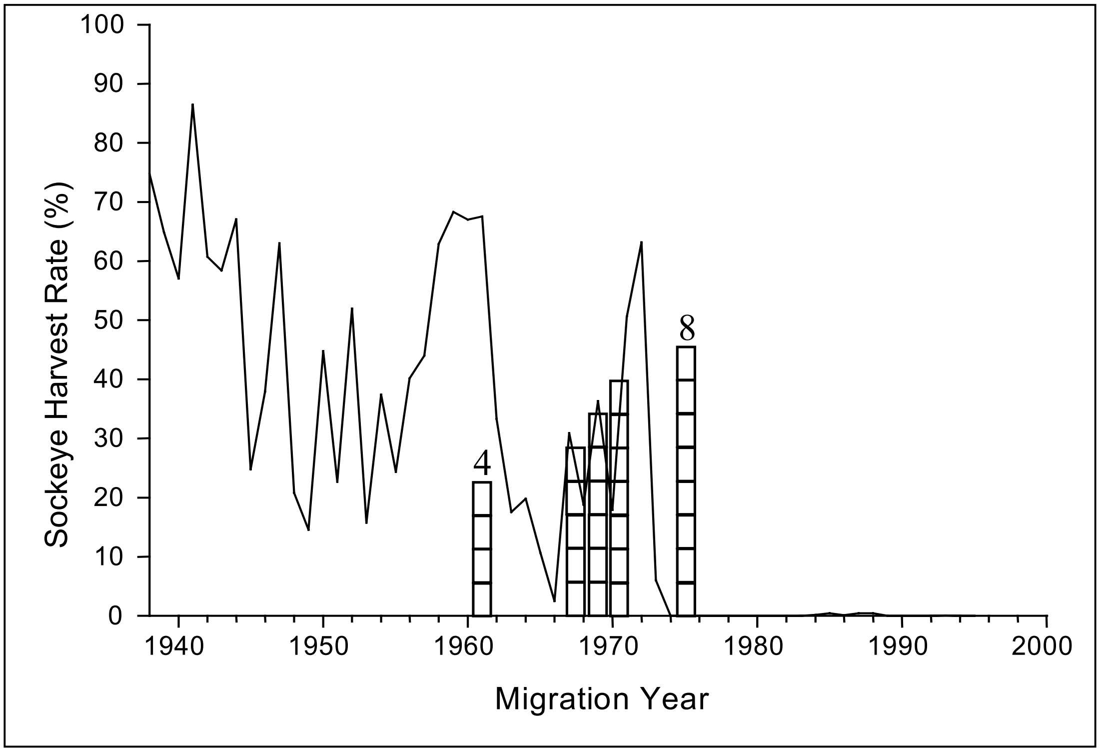
Sources: Counts at Redfish Lake from Kiefer et al. (1991). Counts at Ice Harbor Dam from ODFW and WDFW (1998).

Figure 7-1. Escapement of Snake River Sockeye Salmon to the Weir at the Outlet from Redfish Lake and to Ice Harbor Dam

7.2.2.1 Per-Project Mortality Rates

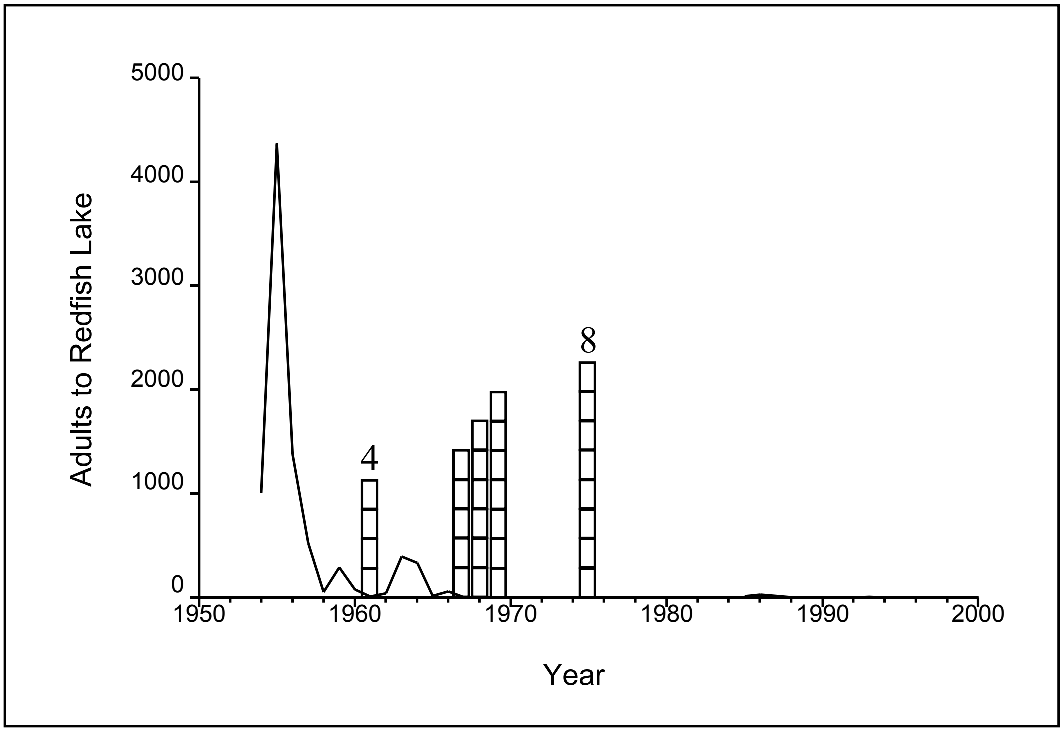
Redfish Lake spawner counts declined steeply from 1955 through 1966, the period during which the number of hydroelectric projects on the mainstem doubled from three to six (Figure 7-3). Although development of the mainstem hydrosystem coincided in time with other factors affecting the survival of Snake River sockeye salmon, it is reasonable to consider the hydrosystem a source of adult loss during migration.

Using conversion rate calculations based on dam counts (Section 4.2.2), Ross (1995) estimated a 15.4 percent 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 7.1), the dam counts, and, therefore, conversion rate estimates for this species, probably include wild residuals and anadromous kokanee.



Note: Figure also provides cumulative number of mainstem (lower Snake and lower Columbia rivers) dams. Harvest rates are calculated as the proportion of the run to Bonneville Dam (ODFW and WDFW, 1998).

Figure 7-2. Mainstem Harvest Rates for Snake River Sockeye Salmon in Zones 1 through 6



Note: Figure also provides cumulative number of mainstem (lower Snake and lower Columbia river) dams
Source: Kiefer et al. (1991)

Figure 7-3. Escapement of Snake River Sockeye Salmon to the Weir at the Outlet from Redfish Lake

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 percent over the four-dam reach between Bonneville and McNary Dams (L. Stuehrenberg, NMFS Northwest Fisheries Science Center, personal communication, December 17, 1998).

$$(1 - 0.11)^{1/4} = 0.97 \text{ (97 percent per-project survival)}$$

$$(1 - 0.97) = 0.03 \text{ (3 percent per-project mortality)}$$

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, then data from the 1997 radio-telemetry study indicate a 22 percent loss through the eight-dam hydrosystem between Bonneville and Lower Granite Dams.

- 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.
- 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:

$$(0.97)^8 = 0.78 \text{ (78 percent system survival)}$$

$$(1 - 0.78) = 0.22 \text{ (22 percent system mortality)}$$

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 Columbia River Power System (because survival appears to be slightly lower in the lower Snake River; Table 7-1).

Table 7-1. 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

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

Source: C. Ross, Fishery Biologist, NMFS, pers. comm., February 23, 1999.

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

7.2.2.2 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 percent passage dates) for adult sockeye salmon between Bonneville and McNary dams to flow (mean daily discharge during June and July) from 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. (1977), these fish are likely to be a mixture of sockeye salmon from the Snake River and the 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.

7.2.2.3 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 decline in sockeye salmon 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.

Currently, 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) purchased the water right using BPA funds. In addition, the U.S. Bureau of Reclamation (BOR) 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 secondary to agricultural water rights, allowing a water user to shut off the fish bypass when the primary 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 high, and an adequate amount of spawning habitat is available (T. Flagg, NMFS representative to the SBTOC, personal communication, 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.

7.2.2.4 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, IDFG, the Shoshone-Bannock Tribe, 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.

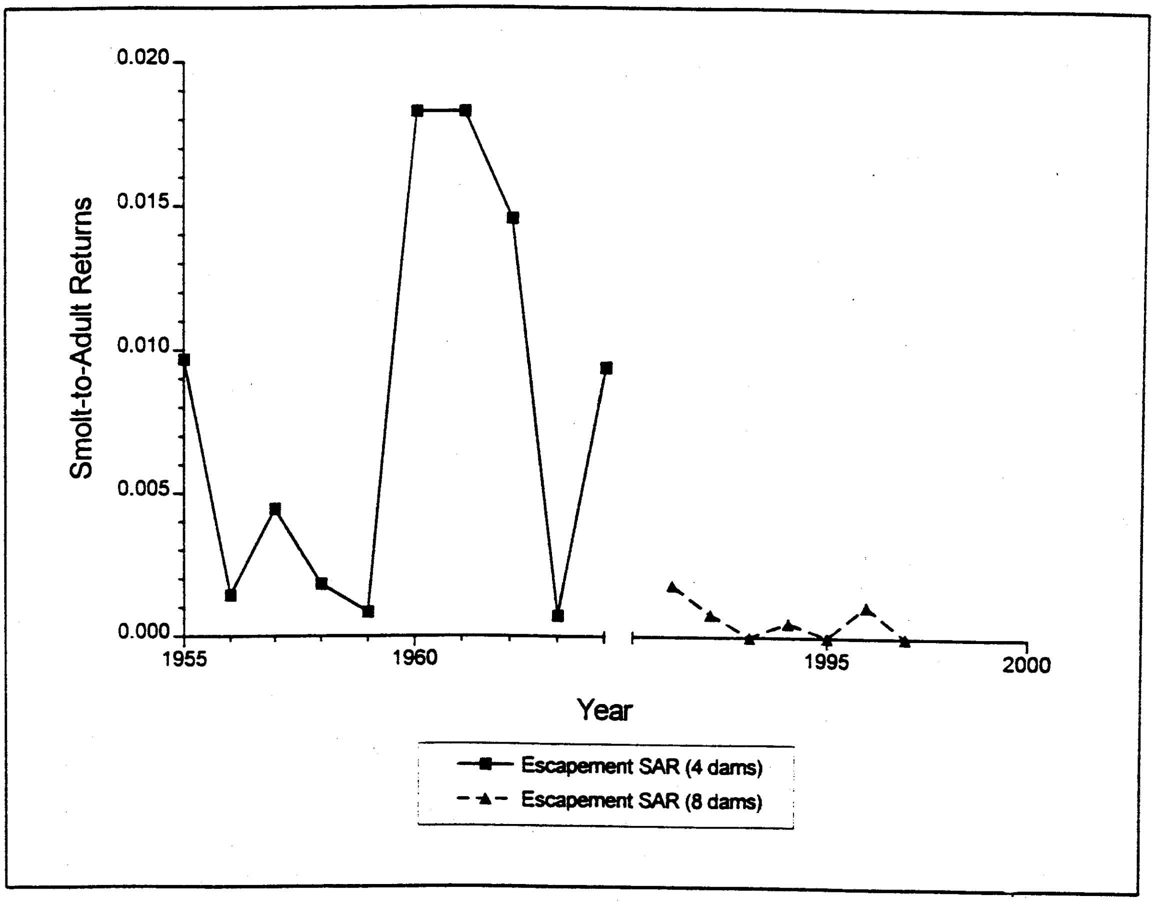
The progeny of captively reared adults are then released to supplement wild populations. The purpose of the program is to maintain the species and prevent extinction in the short term and to jump start the reestablishment of sockeye salmon runs to the waters of the Stanley Basin in the long term. Ultimately, 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.

7.3 Egg-to-Smolt Stage

During 1998, NMFS and IDFG released approximately 160,000 sub-yearling 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.

7.4 Smolt-to-Adult Return Stage

The number of hydroelectric projects on the mainstem doubled from three to six from 1960 through 1969. SAR rates from 1955 through 1964 averaged 0.8 percent (Bjornn et al., 1968). From 1991 through 1996, average SAR declined by over 90 percent to 0.07 percent (C Petrosky, Fishery Biologist, IDFG, personal communication, December 21, 1998) (Figure 7-4). These SARs represent the survival rates of wild residual smolts from Redfish Lake that have returned as adults ("escapement SAR," as defined in NMFS [1998]).



Note: Data were taken during periods when up to four and between five and eight mainstem dams were in place. SARs include escapement (but not harvest). Source: P. Kline (pers. comm.); Bjornn et al. (1968).

Figure 7-4. Smolt-to-Adult Return Rates for Snake River Sockeye Salmon

Table 7-2. Fish Guidance Efficiencies for Sockeye Salmon at John Day and McNary Dams

Project	Test Dates	Screen Type	Op. Gate Position	Average Percent Fish Guidance Efficiency Sockeye Salmon	Yearling Chinook	Steel-head	Subyearling Chinook	No. Observ.
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	ELBS 3/	NOG	Slot7B-80%	83%	95%	—	n = 16

1/ LSTS = lowered submerged traveling screen
2/ STS = submerged traveling screen
3/ ELBS = extended-length submerged bar screen
4/ ROG = raised operating gate
5/ NOG = no operating gate

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.

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 7.2.2.3). 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.

7.4.1 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 from 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 FGEs for sockeye salmon at mainstem dams indicate that, where submerged traveling 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 were used did sockeye salmon guidance rise to that of spring/summer chinook salmon (Table 7-2).

Descaling rates for sockeye salmon at lower Snake River dams and McNary Dam may indicate a mechanism for increased mortality resulting from dam passage. Descaling rates for the period 1981 through 1997 are shown in Table 7-3. These data, when compared with similar estimates for steelhead and spring/summer chinook salmon (Marmorek et al., 1998), 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., 1998). 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. However, data linking these higher descaling rates to higher mortality are totally lacking.

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 nontransported fish. Transport-survival studies for sockeye salmon trucked from Priest Rapids Dam were performed from 1984 through 1988. However, Chapman et al. (1997) reviewed these studies and concluded that the protocols were specific to the mid-Columbia reach and that 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 for juvenile Snake River sockeye salmon migrating through either the mainstem Snake or Columbia River. Zimmerman (1997) reported that approximately 85 percent of the identifiable fish in the guts of northern pikeminnow from lower Snake River reservoirs were salmonids. Of these, 50 percent could not be identified by 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.

7.5 Effects of Ocean and Estuarine Conditions

Survival through the estuary and ocean life-history phase is affected by year-to-year variation and multiyear 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 ESU. Therefore, it is not possible to predict the degree to which changes in ocean conditions have influenced the decline of this ESU 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.

7.6 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 ESU (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 present. The only Snake River hatchery program is the emergency captive broodstock for Redfish Lake; although this program entails genetic and other risks to this ESU, these risks are considered to be lower 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 (Scientific Review Team [SRT] and Independent Scientific Advisory Board [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 SBTOC is not likely to continue the captive broodstock program indefinitely if ongoing sources of mortality elsewhere in the life cycle are not reversed.

Table 7-3. 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 stock	9.9	0	13.9	9.7	
	Wild stock	24.5	10.7	14.1	18.7	1/, 3/
1996	Hatchery stock	3.8	5.3	6.7	11.6	
	Wild stock	18.4	14.8	5.9	11.5	3/
1995	Hatchery stock	3.2	9.4	4.8	5.7	
	Wild stock	30.1	15.7	13.6	18.3	3/
1994	Hatchery stock				7.8	
	Wild stock	12.5	15.1	21.0	12.4	2/, 3/
1993	Hatchery stock			26.6	2.9	
	Wild stock	27.3	11.1		8.5	3/
1992	Combined	2.3	6.6		13.1	4/
1991	Combined	0.5	5.9		10.8	
1990	Combined		10.0			
1989	Combined				16.8	
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 facilities (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 through 1997 reported in annual reports of the Juvenile Fish Transportation Program. Numerous authors. U.S. Army Corps of Engineers, 1995 through 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 8 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]).

7.7 Relevance to the Analysis of Hydrosystem Management Alternatives

Waiting for further research on the passage survival of Snake River sockeye salmon is not an option. 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 the following questions:

- 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 inriver migrants for both groups (and how do these vary with inriver conditions and inriver migration routes)?

Because the various life-history forms are not distinguished in the existing literature, it is impossible to 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 6.0. 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.

8. A Cumulative Risk Analysis

All preceding quantitative discussion has relied heavily on the interpretation of results from PATH. To complement PATH, NMFS has undertaken an additional analytical approach referred to as the CRI. Unlike PATH, CRI does not rely on large, detailed models, but rather is a chain of connected logical steps, each step simpler and easier to understand than the richly detailed PATH models. While the PATH models offer a great deal in terms of careful treatment of hydrosystem passage, the same models carry with them the cost of being so unwieldy and difficult to document that it would be difficult for any external scientist to repeat or duplicate the analyses. In designing this complementary CRI approach, NMFS sought to address four shortcomings of PATH analyses:

1. PATH does not provide an estimate of the risk of extinction for any index populations; an estimate of this risk is an important piece of information for decision-making. In particular, decision makers need to know the potential costs of delaying action.
2. The performance measures suggested by NMFS in its 1995 Biological Opinion and subsequently used by PATH (described in Section 2.2.1) are difficult to interpret; although PATH's performance measures depend on population numbers and population growth, the connection is not transparent.
3. The PATH models were initially designed to provide detailed analyses of different fish passage scenarios. Although a number of sensitivity analyses were performed to examine other Hs (harvest, habitat, and hatcheries), the analyses do not lend themselves well to comparison among Hs in a common currency and on common footing.
4. In its thoroughness, PATH investigated an enormous diversity of hypotheses and assumptions; the cost of this inclusiveness is that certain fundamental comparisons and examinations are lost in its complexity.

The CRI approach cannot replace PATH's detailed examination of modifications of transport or fish-passage systems, and is not intended to do so. Rather, the CRI offers a more simplified approach to help make informed decisions about management options. Like PATH, the CRI also has shortcomings, and these shortcomings are summarized in Section 8.6.

8.1 Overview of CRI Analyses

In lieu of a complex of models with several hundred parameters that need to be specified in order to generate predictions, the CRI breaks the analyses into six steps:

1. Estimate the population growth rate for index stocks and entire ESUs. Then, using these estimates of population growth rate, estimate the risk of substantial decline and extinction for those stocks and ESUs.
2. Construct demographic projection matrices that depict current demographic performance rates.
3. Perform sensitivity analyses to assess where in the life cycles of salmonids there exist the greatest opportunities for promoting recovery, as measured by changes in the annual population growth rate (or dominant eigenvalue associated with each matrix).

4. Manipulate the values in baseline matrices to represent hypothesized demographic responses to management actions for which a population response is known, and calculate the percent increase in annual population growth rate associated with each management action. Determine whether the change in annual population growth rate is sufficient to produce a stable or growing population (rather than a decreasing one).
5. Explore whether the connection between the management action and the hypothesized demographic response is biologically feasible or those management actions that seem numerically effective are possible.

In addition, so that others can repeat analyses or perform alternative analyses, all data used in analyses and examples of analyses are placed on a public website.

A major philosophical difference between CRI and PATH analyses is that CRI separates sensitivity analyses and numerical experiments concerning management scenarios from the question of what is biologically feasible. This approach better draws attention to what data gaps exist and makes the key questions more transparent.

In addition to the above general issues, the CRI approach differs from the PATH analyses in specific technical ways. First, in the absence of statistical evidence to the contrary, the CRI analyses are density independent, whereas all PATH models start with the assumption that a Ricker function describes recruits per spawner. Most PATH analyses focus on deviations from the Ricker fit and possible explanations for patterns in those deviations. Density dependence must play a role in salmonid population dynamics, but CRI regression analyses generally fail to find evidence supporting density-dependent recruitment when population data from 1980 onward are analyzed on a stock-by-stock basis. This does not mean that NMFS rejects the notions of carrying capacity or density dependence. Rather, NMFS suggests that when analyzing scenarios with respect to viability, calculations of extinction risk are best done with density-independent models, unless there are data that strongly support inclusion of density effects. The result of this difference is that CRI projections are less optimistic than PATH projections because in PATH simulations, populations benefit from a boost in recruitment rates as numbers decline. CRI analyses do not assume this effect, and the populations experience no increase in recruitment as the number of spawners decreases.

Second, the performance measures for the CRI analyses are average annual rates of population change and probabilities of decline and extinction, whereas the performance measures for PATH are less direct. This difference is especially striking with respect to discussions of spring/summer chinook salmon. In the PATH 1998 report, it is difficult to find direct estimates of population sizes, population growth rates, or probabilities of extinction. The PATH 1999 draft report for fall chinook salmon does report some results in terms of expected numbers of fish, which are easier to interpret than the survival and recovery standards typically relied on by earlier PATH analyses.

Third, CRI does not explicitly include the mathematical constructs of extra mortality or differential delayed mortality in matrix analyses. Instead, as described above, the CRI relies on an average demographic matrix that estimates population growth under current conditions. Then, using this baseline matrix, simulations are run to see how different alterations of stage-specific demography (including stages at which extra mortality would be expressed) influence annual rates of population change. Finally, there is discussion of the feasibility of obtaining particular demographic improvements with particular management actions. Thus, instead of examining a complicated assemblage of models involving extra mortality and differential delayed mortality as potential explanations of unexplained residual variation, the CRI matrix simply simulates the effect of improving survival during downstream migration, and survival below Bonneville Dam. A separate step in the CRI analysis asks what data exist to support the conclusion that these survival improvements could, in reality, be realized by dam breaching (or other management options). This makes more transparent the importance of the question whether fish suffer a latent mortality due to the presence of the hydrosystem, but which is not directly observed during downstream or upstream passage.

8.2 Estimating Population Growth Rates and Risks to Populations

NMFS conducted a standardized, quantitative risk analysis applied to 11 of the 12 salmonid ESUs in the Columbia Basin that have protection under the ESA (McClure et al., in review). This analysis, which includes 8 ESUs outside of the Snake River Basin, is described below. In addition to these listed ESUs, several "healthy" stocks for comparison (Hanford Reach fall chinook and three stocks belonging to the Washington Coastal chinook ESU) were included for comparative purposes. The inclusion of such "control groups" can provide a substantive basis for interpreting the status of more imperiled populations. The Snake River sockeye were excluded from analysis, because this ESU is maintained in a captive broodstock program.

In this standardized analysis, NMFS used diffusion approximation methods to address three sequential questions: 1) What is the rate of population change? 2) What is the risk of extinction or severe decline for each stock given current conditions? 3) How much improvement in the rate of population change is needed to avoid extinction or severe decline? Although a complete viability analyses will consider other factors (such as genetic diversity) in addition to these strictly demographic ones (Soule and Gilpin, 1986), these demographic analyses are a critical first step towards a complete viability analysis, and are often the only analyses that currently available data support. In addition, NMFS determined the range of those population parameters and risk estimates, given the potential for in-stream reproduction by hatchery fish to mask the true population trajectory.

8.2.1 Methods for Estimating Population Growth Rate and Risks

8.2.1.1 Time Period Analyzed

The analysis was restricted to the years since 1980 in order to determine the status of stocks and risks the stocks face under current conditions. Changes to the hydropower system were a main component of this choice, since prior to that time, the hydropower system on the Columbia River was in a state of flux. The final dam on the mainstem Columbia was completed in 1971, the last of the four lower Snake River dams was completed in 1975, and the full complement of turbines installed by 1979. Additional major engineering changes to the lower Snake River dams and other mainstem dams on the Columbia river were completed by the early 1980s. In addition, the reservoir storage capacity in the Columbia was nearly doubled in 1975, when the Libby and Mica Dams were completed. Including data from years prior to 1980 would therefore confound any evaluation of current status by implicitly incorporating conditions that are no longer present. In addition, the quality of early data is not uniform across ESUs (Zabel and Williams, 2000). By using more recent data (i.e., the 1980 to the present time period) McClure et al. (in review) eliminated some, though certainly not all, problems with differences in data quality among ESUs.

8.2.1.2 Data Used in Analyses

Determining population growth rates and associated risks required stock-level time series of fish abundance or density, age structure, and the proportion of hatchery spawners. Spawner abundance data consisted of either direct counts of returning adults at dams or weirs, index counts (such as the density of redds, the gravel nests made by spawning females), or estimates of total spawners. At the ESU-level, dam counts encompassing the entire ESU were available for Snake River steelhead, fall chinook and spring/summer chinook, Upper Columbia spring chinook and steelhead, and Upper Willamette chinook and steelhead. For the other four ESUs, an ESU-level count was approximated by aggregating all stocks within that ESU for which there was a total live spawner time series. In order to best represent the number of fish on the spawning grounds, fish from the time series that were harvested in-river or taken into hatcheries upstream from the dam counts were subtracted. Age structure at the stock level was used when available; otherwise, estimates of age structure for the entire ESU were applied to all stocks within the ESU. Estimates of the proportion of hatchery-origin spawners were available for approximately two-thirds of the stocks analyzed; about half of these were point estimates rather than time series. These estimates of the proportion of hatchery fish on the spawning grounds were based either on direct observations of fin-clipped fish or were derived from estimates of hatchery stray rates. When no estimate of the proportion of hatchery and wild spawners was available, population growth rate was calculated for the cumulative spawner counts, which include both wild and hatchery born spawners.

8.2.1.3 Estimating Population-level Parameters

Spawner time series were used to estimate population growth rate and risks by fitting a stochastic exponential decline model to the data and then using diffusion approximation methods (Dennis et al., 1991) to estimate risks. Previously developed parameter estimation methods were not appropriate for raw spawner counts for several reasons. First, spawner counts can be problematic because they represent only a single life stage and are therefore not a representative sample of the entire population. In addition, because salmon return to freshwater several years after eggs are laid, they are prone to boom and bust cycles in annual spawner numbers. These cycles confound parameter estimation. Second, sampling error is likely to be very high in spawner count data (Hilborn et al., 1999). Large sampling error results in overestimates of the environmental variance, which lead to correspondingly poor estimates of any risk metrics that incorporate this measure of variance (Holmes, in press). Third, the regular introduction of reproducing hatchery-origin spawners (in effect, fish from another population) confounds the parameter estimates of the instantaneous rate of population growth for the wild population. The modified parameter estimation methods used are robust to sampling error and allowed incorporation of the input of hatchery-origin spawners (Holmes, in review).

Weighting spawner counts—The methods developed by Holmes (in press) require the use of a running sum that functions to filter out sampling error and age-structure cycles. The parameter estimates are not particularly sensitive to the structure of the running sum as long as it is not too long (Holmes, in press). Thus, a weighted running sum was developed that served the dual purpose of filtering the data and providing an estimate of the total living current or future spawners. The total living current or future spawners is a population size estimate which can then be used for extinction analyses. To generate this running sum, R_t, the estimated number of future spawners, $SS*S_t$, was weighted by the mean age at which fish return to the spawning ground to estimate those individual fish alive at time t that are now spawning or will live to spawn in the future:

$$R_t \approx SS \sum_{age=1}^{\text{max age}} \varphi_j S_{t-j} \qquad [8\text{-}1]$$

where SS is the mean number of future spawners produced by current spawners and φ_j is the average fraction of fish of age j that have yet to spawn or are spawning this year. φ_j is related to the average distribution of return ages as follows:

$$\begin{aligned} \varphi_1 &= 1 && \mathrm{j} = 1 \\ \varphi_j &= 1 - \sum_{age=1}^{j-1} D_i && \mathrm{j} > 1 \end{aligned} \qquad [8\text{-}2]$$

where D_i is the fraction of spawners that are age i.

These transformed spawner counts were tested for their fit to the assumptions of the underlying stochastic process: 1) the relationship between the variance and the lag in $\ln(R_{t+\tau}/R_t)$ is linear, using the R^2 of a least-squares fit through the variance data (Figure 8-1); 2) $\ln(R_{t+1}/R_t)$ is distributed normally and there are not significant outliers (using the dffits statistic > 2); 3) density-dependent processes do not occur (following Dennis and Taper, 1994); 4) there are no temporal trends in σ (using a method analogous to Dennis and Taper's test for density-dependence); and 5) there is no significant serial autocorrelation in the R_{t+1}/R_t ratios (by de-trending the ratios and using Spearman's rank correlation test). All tests were done at the $p < 0.05$ significance level with no adjustment for the fact that 110 tests were done. There was a good fit to all assumptions with the following exceptions: at the ESU-level, the Upper Columbia spring chinook, Lower Columbia chinook, Lower Columbia steelhead and Mid-Columbia steelhead time series exhibit a downward trend in $R_{t+\tau}/R_t$ ratios, as do several stocks within the Snake River spring/summer chinook ESU. It should be kept in mind that simulations (Shenk et al., 1998) indicate that significant trends appear by chance 25 to 30 percent of the time in stochastic age-structured processes. Several stocks show evidence of density depensatory or compensatory processes (Table 8-1). σ (and consequently extinction risks) will be underestimated when there is depensatory density dependence or declining trends in $R_{t+\tau}/R_t$ ratios. A handful of stocks showed evidence of 1st order autocorrelation in $R_{t+\tau}/R_t$ ratios. When autocorrelation is present, σ^2 is underestimated using our methods but σ should be unaffected (Tuljapurkar, 1989).

The parameter estimation methods and tests of their performance are discussed in detail by Holmes (in press).

Estimating the instantaneous rate of change—The running sum method was used to estimate the mean instantaneous rate of change, μ, for each stock and ESU:

$$\hat{\mu}_{run} = \text{mean}(\ln(R_{t+1} / R_t)) \qquad [8\text{-}3]$$

This method gives an estimate of μ that is resistant to severe age-structure perturbations (Holmes, in press).

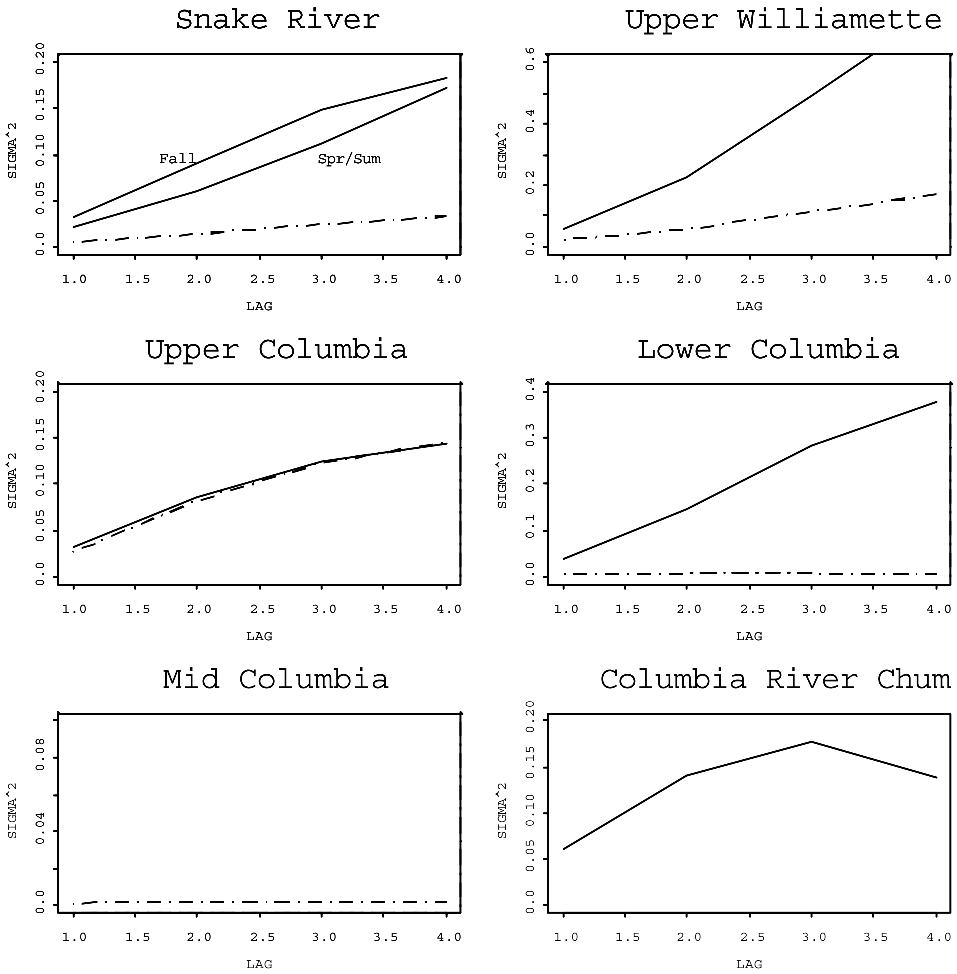


Figure 8-1. Recruits Per Spawner Versus Spawner Density for Spring/Summer Chinook Salmon Index Stocks

The variance in ln($R_{t+\tau}/R_t$) where R_t is the weighted sum of spawner counts as described in the text. Plots for steelhead are dashed; plots for chinook are solid. A basic assumption of the σ^2 parameter estimation is that this relationship is approximately linear. The slope of the σ^2 versus t line is used to estimate the variance in μ due to environmental stochasticity. Plots that are flat indicate ESUs for which the variance was 0 or close to zero.

Table 8-1. Summary of Estimated Population Size (Wild Only), Parameter Estimates, Risk of Extinction and 90 Percent Decline in Abundance and Needed Improvements in I to Reduce Risk of Decline or Extinction in 100 Years to Below 5 Percent

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			Population Parameter Estimates					Risk of Extinction			Risk of 90% Decline				
ESU	Stock	Pop. size estimate	μ	σ^2	λ	95% confidence interval low	95% confidence interval up	24 yrs	100 yrs	Req. inc. (%)	24 yrs	100 yrs	Req. inc. (%)	Diag. tests &	Note $
Lower Columbia chinook	**ESU Level**	**NA**	**(0.02)**	**0.12**	**0.98**	**0.78**	**1.24**	**NA**	**NA**	**NA**	**0.12**	**0.42**	**5.0**	**t**	**i**
	Bear Ck	253	(0.14)	0.20	0.87	0.64	1.18	0.21	0.98	19.5	0.68	0.99	22.0	l	
	Big Ck	2,982	(0.02)	0.04	0.98	0.87	1.10	0.00	0.00	0.0	0.03	0.50	3.5		
	Clatskanie	28	(0.07)	0.44	0.93	0.63	1.37	0.48	0.88	28.5	0.42	0.76	22.0		
	Cowlitz Tule	NA	(0.03)	0.10	0.97	0.81	1.17	NA	NA	NA	0.15	0.56	6.0		i
	Elochoman	NA	0.04	0.43	1.04	0.71	1.53	NA	NA	NA	0.15	0.18	9.0		i
	Germany	NA	(0.02)	0.14	0.98	0.78	1.23	NA	NA	NA	0.16	0.48	6.5		i
	Gnat	105	(0.02)	0.45	0.98	0.67	1.46	0.18	0.57	16.0	0.28	0.46	16.0		
	Grays Tule	NA	(0.11)	0.42	0.90	0.62	1.31	NA	NA	NA	0.54	0.91	26.0		i
	Kalama Spr	NA	(0.12)	0.14	0.89	0.71	1.11	NA	NA	NA	0.61	0.99	17.5	d, l	i
	Kalama	NA	0.03	0.52	1.03	0.68	1.57	NA	NA	NA	0.19	0.21	12.0	a	i
	Klaskanine	27	(0.07)	0.77	0.94	0.55	1.60	0.57	0.88	47.5	0.44	0.69	30.5		
	Lewis R Bright	NA	(0.01)	0.04	0.99	0.88	1.12	NA	NA	NA	0.02	0.25	2.5		i
	Lewis Spr	NA	(0.05)	0.42	0.95	0.65	1.38	NA	NA	NA	0.37	0.67	19.0	t, a	i
	Lewis, E Fk Tule	NA	(0.01)	0.02	0.99	0.91	1.08	NA	NA	NA	0.00	0.14	1.0		i
	Lewis and Clark	1	(0.56)	2.61	0.57	0.21	1.54	1.00	1.00	0.0	0.92	1.00	####	t	
	Mill Fall	307	(0.16)	0.18	0.85	0.62	1.16	0.25	1.00	21.5	0.78	1.00	24.5		
	Plympton	2,991	0.00	0.14	1.00	0.80	1.24	0.00	0.04	0.0	0.11	0.29	5.0		
	Sandy Late	4,135	(0.02)	0.01	0.98	0.90	1.08	0.00	0.00	0.0	0.00	0.28	1.5		
	Skamokawa	NA	(0.15)	0.04	0.86	0.77	0.97	NA	NA	NA	0.89	1.00	17.0		i
	Youngs	19	(0.01)	1.04	0.99	0.51	1.91	0.58	0.80	64.5	0.34	0.46	30.0		
Upper Columbia spring chinook	**ESU Level**	**1,872**	**(0.16)**	**0.04**	**0.85**	**0.76**	**0.95**	**0.00**	**1.00**	**13.5**	**0.96**	**1.00**	**19.0**	**t, d, a**	
	Methow	324	(0.14)	0.26	0.87	0.65	1.16	0.24	0.97	22.0	0.67	0.99	24.5	t, d	
	Entiat	159	(0.14)	0.03	0.87	0.79	0.96	0.03	1.00	13.0	0.88	1.00	15.5	t, d	
	Wenatchee	745	(0.22)	0.02	0.81	0.74	0.88	0.03	1.00	19.5	1.00	1.00	24.5	t, d	

Table 8-1. Summary of Estimated Population Size (Wild Only), Parameter Estimates, Risk of Extinction and 90 Percent Decline in Abundance and Needed Improvements in I to Reduce Risk of Decline or Extinction in 100 Years to Below 5 Percent

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			Population Parameter Estimates					**Risk of Extinction**			**Risk of 90% Decline**				
ESU	**Stock**	**Pop. size estimate**	**μ**	**σ^2**	**λ**	**95% confidence interval low**	**up**	**24 yrs**	**100 yrs**	**Req. inc. (%)**	**24 yrs**	**100 yrs**	**Req. inc. (%)**	**Diag. tests &**	**Note $**
Snake River spring/summer chinook	**ESU Level**	**23,336**	**(0.04)**	**0.01**	**0.96**	**0.91**	**1.02**	**0.00**	**0.00**	**0.0**	**0.00**	**0.91**	**3.5**		
	Bear Ck	736	0.02	0.15	1.02	0.83	1.25	0.00	0.03	0.0	0.07	0.15	3.0	t	
	Imnaha R	657	(0.08)	0.04	0.93	0.83	1.03	0.00	0.78	5.5	0.33	1.00	9.5		
	Johnson Ck	457	0.01	0.05	1.01	0.90	1.14	0.00	0.00	0.0	0.01	0.07	0.5		
	Marsh Ck	291	(0.01)	0.13	0.99	0.82	1.19	0.00	0.19	3.0	0.13	0.39	5.5	a	
	Minam R	338	(0.01)	0.16	0.99	0.80	1.23	0.00	0.17	3.0	0.13	0.33	5.5		
	Poverty Ck	1,051	0.01	0.08	1.01	0.86	1.17	0.00	0.01	0.0	0.04	0.16	2.0	t	
	Sulphur Ck	207	0.04	0.41	1.04	0.74	1.47	0.05	0.21	7.0	0.15	0.17	8.5	t	
	Alturas Lake Ck	NA	(0.29)	0.08	0.75	0.64	0.87	NA	NA	NA	1.00	1.00	37.5		i, h
	American R	NA	(0.10)	0.26	0.91	0.69	1.19	NA	NA	NA	0.50	0.92	19.0		i, h
	Bear Valley Ck	NA	(0.01)	0.09	0.99	0.84	1.17	NA	NA	NA	0.08	0.30	4.0		i, h
	Big Sheep Ck	NA	(0.08)	0.47	0.93	0.62	1.38	NA	NA	NA	0.45	0.78	23.5		i, h
	Beaver Cr	NA	(0.05)	0.11	0.95	0.79	1.15	NA	NA	NA	0.24	0.78	8.5		i, h
	Bushy Fork	NA	(0.02)	0.03	0.98	0.90	1.07	NA	NA	NA	0.01	0.43	2.5		i, h
	Camas Cr	NA	(0.08)	0.09	0.92	0.77	1.10	NA	NA	NA	0.41	0.97	11.5		i, h
	Cape Horn Cr	NA	0.05	0.13	1.05	0.85	1.30	NA	NA	NA	0.02	0.03	0.0	d	i, h
	Catherine Ck	NA	(0.07)	0.12	0.93	0.76	1.14	NA	NA	NA	0.37	0.92	11.5	t	i, h
	Crooked Fork	NA	0.00	0.04	1.00	0.90	1.11	NA	NA	NA	0.01	0.13	1.0		i, h
	Elk Ck	NA	0.05	0.25	1.05	0.80	1.38	NA	NA	NA	0.08	0.09	2.5	t	i, h
	Grande Ronde R	NA	(0.05)	0.06	0.95	0.83	1.10	NA	NA	NA	0.18	0.86	7.0	t	i, h
	Knapp Cr	NA	(0.12)	0.16	0.89	0.71	1.12	NA	NA	NA	0.60	0.99	18.0	d	i, h
	Lake Cr	NA	0.06	0.06	1.06	0.91	1.23	NA	NA	NA	0.00	0.00	0.0	t, d	i, h
	Lemhi R	NA	(0.02)	0.25	0.98	0.75	1.28	NA	NA	NA	0.24	0.51	10.5	a	i, h
	Loon Ck	NA	0.00	0.02	1.00	0.94	1.07	NA	NA	NA	0.00	0.01	0.0		i, h
	Lostine Ck	NA	(0.05)	0.06	0.95	0.83	1.10	NA	NA	NA	0.16	0.85	7.0		i, h
	Lower Salmon R	NA	(0.09)	0.08	0.92	0.79	1.07	NA	NA	NA	0.44	0.99	12.0	a	i, h
	Lower Valley Ck	NA	(0.08)	0.15	0.92	0.75	1.14	NA	NA	NA	0.42	0.93	13.5	a	i, h
	Moose Ck	NA	(0.06)	0.05	0.94	0.84	1.06	NA	NA	NA	0.20	0.94	7.5		i, h
	Newsome Ck	NA	0.03	0.05	1.03	0.91	1.16	NA	NA	NA	0.00	0.01	0.0		i, h

Table 8-1. Summary of Estimated Population Size (Wild Only), Parameter Estimates, Risk of Extinction and 90 Percent Decline in Abundance and Needed Improvements in I to Reduce Risk of Decline or Extinction in 100 Years to Below 5 Percent

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			Population Parameter Estimates					**Risk of Extinction**			**Risk of 90% Decline**				
						95% confidence interval									
ESU	**Stock**	**Pop. size estimate**	**μ**	**σ^2**	**λ**	**low**	**up**	**24 yrs**	**100 yrs**	**Req. inc. (%)**	**24 yrs**	**100 yrs**	**Req. inc. (%)**	**Diag. tests &**	**Note $**
	Red R	NA	(0.10)	0.03	0.91	0.83	1.00	NA	NA	NA	0.49	1.00	10.5		i, h
	Salmon R E Fk	NA	(0.06)	0.26	0.94	0.71	1.24	NA	NA	NA	0.37	0.78	15.0	t, a	i, h
	Salmon R S Fk	NA	0.06	0.09	1.06	0.90	1.25	NA	NA	NA	0.01	0.01	0.0	t, d	i, h
	Secesh R	NA	(0.02)	0.00	0.98	0.95	1.01	NA	NA	NA	0.00	0.57	1.5	t	i, h
	Selway R	NA	(0.09)	0.01	0.91	0.86	0.97	NA	NA	NA	0.40	1.00	9.5		i, h
	Upper Big Ck	NA	(0.03)	0.10	0.97	0.82	1.15	NA	NA	NA	0.16	0.62	6.5	t	i, h
	Upper Salmon R	NA	(0.10)	0.04	0.90	0.82	1.00	NA	NA	NA	0.55	1.00	11.5		i, h
	Upper Valley Ck	NA	0.03	0.61	1.03	0.68	1.57	NA	NA	NA	0.21	0.24	14.5	a	i, h
	Wallowa Ck	NA	(0.15)	0.49	0.86	0.57	1.29	NA	NA	NA	0.65	0.97	34.0		i, h
	Wenaha R	NA	0.00	0.10	1.00	0.83	1.21	NA	NA	NA	0.07	0.24	3.5	t	i, h
	Whitecap Ck	NA	(0.10)	0.06	0.90	0.78	1.05	NA	NA	NA	0.55	1.00	13.0	t	i, h
	Yankee Fork	NA	(0.12)	0.18	0.88	0.69	1.13	NA	NA	NA	0.63	0.99	19.5		i, h
	Yankee West Fk	NA	(0.01)	0.18	0.99	0.77	1.27	NA	NA	NA	0.16	0.39	7.0		i, h
Snake River Basin Chinook	**ESU Level**	**1,505**	**(0.06)**	**0.05**	**0.94**	**0.81**	**1.09**	**0.00**	**0.40**	**3.5**	**0.24**	**0.96**	**8.5**		
	Snake R Basin	1,505	(0.06)	0.05	0.94	0.81	1.09	0.00	0.40	3.5	0.24	0.96	8.5		
Upper Willamette Chinook	**ESU Level**	**6,859**	**0.01**	**0.24**	**1.01**	**0.76**	**1.34**	**0.00**	**0.05**	**0.0**	**0.15**	**0.26**	**6.5**		
	McKenzie R.	4,704	0.03	0.21	1.03	0.79	1.34	0.00	0.01	0.0	0.09	0.12	3.0	t	
Upper Columbia fall Chinook	Hanford Reach	163,868	(0.01)	0.07	0.99	0.85	1.17	0.00	0.00	0.0	0.04	0.24	2.5		h
WA Coast Chinook	Hoh R Fall	11,900	0.03	0.01	1.03	0.95	1.13	0.00	0.00	0.0	0.00	0.00	0.0		h
	Queets R Fall	12,879	0.11	0.01	1.12	1.01	1.23	0.00	0.00	0.0	0.00	0.00	0.0		h
	Willapa R Fall	15,651	0.07	0.01	1.07	0.98	1.16	0.00	0.00	0.0	0.00	0.00	0.0	l	h

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			Population Parameter Estimates					Risk of Extinction			Risk of 90% Decline				
ESU	Stock	Pop. size estimate	μ	σ^2	λ	95% confidence interval low	up	24 yrs	100 yrs	Req. inc. (%)	24 yrs	100 yrs	Req. inc. (%)	Diag. tests &	Note $
Columbia River chum	**ESU Level**	**NA**	**0.03**	**0.03**	**1.04**	**0.94**	**1.13**	**NA**	**NA**	**NA**	**0.00**	**0.00**	**0.0**	**l**	**i**
	Grays R west fk	NA	0.21	0.20	1.23	0.96	1.59	NA	NA	NA	0.00	0.00	0.0	l	i
	Grays R	NA	(0.04)	0.12	0.96	0.78	1.16	NA	NA	NA	0.24	0.73	8.5		i
	Hardy Ck	NA	0.04	0.06	1.05	0.92	1.19	NA	NA	NA	0.00	0.00	0.0	l	i
	Crazy J	NA	0.15	0.03	1.16	1.05	1.28	NA	NA	NA	0.00	0.00	0.0	a	i
	Hamilton	NA	(0.08)	0.05	0.92	0.81	1.05	NA	NA	NA	0.40	1.00	10.5	l	i
	Hamilton Sprs	NA	0.11	0.59	1.11	0.74	1.68	NA	NA	NA	0.10	0.10	6.0	t*	i
Lower Columbia steelhead	**ESU Level**	**NA**	**(0.02)**	**0.00**	**0.98**	**0.97**	**0.98**	**NA**	**NA**	**NA**	**0.00**	**0.96**	**0.5**	**t**	**i**
	Clackamas Sum	2,720	(0.11)	0.01	0.89	0.84	0.96	0.00	1.00	5.5	0.77	1.00	11.5	t	
	Clackamas Win	937	(0.04)	0.00	0.96	0.92	1.00	0.00	0.00	0.0	0.00	1.00	3.0		
	Green R Win	660	(0.10)	0.21	0.90	0.58	1.41	0.06	0.86	14.0	0.53	0.96	18.0	t	
	Kalama R Sum	5,902	0.03	0.03	1.04	0.93	1.16	0.00	0.00	0.0	0.00	0.00	0.0	d	
	Kalama R Win	4,228	0.01	0.01	1.01	0.95	1.06	0.00	0.00	0.0	0.00	0.00	0.0		
	Sandy Win	3,471	(0.06)	0.03	0.94	0.85	1.04	0.00	0.09	0.5	0.13	0.98	6.5		
	Toutle Win	3,008	(0.13)	0.00	0.88	0.86	0.89	0.00	1.00	6.0	1.00	1.00	12.5		
Mid Columbia steelhead	**ESU Level**	**NA**	**(0.13)**	**0.00**	**0.88**	**0.88**	**0.88**	**NA**	**NA**	**NA**	**1.00**	**1.00**	**11.0**	**t**	**i**
	Deschutes R Sum	9,157	(0.15)	0.00	0.86	0.83	0.90	0.00	1.00	7.0	1.00	1.00	14.5		
	Mill Ck Sum	NA	(0.01)	0.05	0.99	0.84	1.17	NA	NA	NA	0.03	0.24	2.5		h
	Shitike Ck Sum	NA	(0.08)	0.01	0.93	0.88	0.98	NA	NA	NA	0.14	1.00	7.5		h
	Warm Spr Nfh Sum	1,031	(0.10)	0.05	0.91	0.76	1.08	0.00	0.92	7.5	0.52	1.00	12.0	t	
	Eightmile Ck Win	NA	(0.11)	1.44	0.90	0.32	2.54	NA	NA	NA	0.52	0.76	51.5		h
	Ramsey Ck Win	NA	0.00	0.38	1.00	0.59	1.71	NA	NA	NA	0.22	0.34	11.5		h
	Fifteen Mile Ck Win	NA	(0.10)	0.05	0.91	0.77	1.07	NA	NA	NA	0.51	1.00	11.5		h
	Umtilla R Sum	5,867	(0.11)	0.00	0.90	0.85	0.94	0.00	1.00	3.5	0.91	1.00	10.5	t, d	
	Yakima R Sum	5,213	0.04	0.02	1.04	0.94	1.16	0.00	0.00	0.0	0.00	0.00	0.0		

Table 8-1. Summary of Estimated Population Size (Wild Only), Parameter Estimates, Risk of Extinction and 90 Percent Decline in Abundance and Needed Improvements in l to Reduce Risk of Decline or Extinction in 100 Years to Below 5 Percent

			Population Parameter Estimates					**Risk of Extinction**			**Risk of 90% Decline**				
						95% confidence interval									
ESU	**Stock**	**Pop. size estimate**	**μ**	**σ^2**	**λ**	**low**	**up**	**24 yrs**	**100 yrs**	**Req. inc. (%)**	**24 yrs**	**100 yrs**	**Req. inc. (%)**	**Diag. tests &**	**Note $**
Upper Columbia steelhead	**ESU Level**	**2,137**	**(0.06)**	**0.04**	**0.94**	**0.83**	**1.07**	**0.00**	**0.25**	**2.5**	**0.19**	**0.97**	**7.5**	**A**	
	Upper Columbia R	2,137	(0.06)	0.04	0.94	0.83	1.07	0.00	0.25	2.5	0.19	0.97	7.5	a	
Snake River Basin steelhead	**ESU Level**	**39,809**	**(0.09)**	**0.01**	**0.91**	**0.86**	**0.97**	**0.00**	**0.13**	**1.0**	**0.48**	**1.00**	**9.5**		
	Snake R A-run	33,603	(0.08)	0.01	0.93	0.87	0.99	0.00	0.01	0.0	0.20	1.00	7.5		
	Snake R B-run	11,833	(0.11)	0.02	0.89	0.81	0.98	0.00	0.93	5.0	0.73	1.00	12.5	t	
Upper Willamette steelhead	**ESU Level**	**10,845**	**(0.06)**	**0.05**	**0.94**	**0.83**	**1.07**	**0.00**	**0.08**	**0.5**	**0.20**	**0.94**	**7.5**		
	Mollala	2,010	(0.05)	0.08	0.95	0.81	1.11	0.00	0.27	3.0	0.23	0.87	8.0		
	N Santiam R	4,690	(0.08)	0.06	0.93	0.81	1.06	0.00	0.40	3.5	0.33	0.99	10.0		
	S Santiam	3,730	(0.03)	0.03	0.97	0.88	1.07	0.00	0.00	0.0	0.03	0.65	4.0		
	Calapooia	416	(0.08)	0.19	0.93	0.72	1.19	0.04	0.74	11.0	0.41	0.88	14.0		

Note: When no hatchery fraction data were available, (noted in comments column) estimates were made using the total (wild + hatchery) spawner count data as described in the text. Estimates are provided for individual stocks and ESUs (in bold).

& Tests for underlying assumptions were made on the running sums of wild spawner only counts where possible; otherwise total mixed counts were used. The codes designate tests that failed at $p < 0.05$. Note that a number of the fails are false-fails because the p-value was not adjusted for 110 tests being conducted. If p value is adjusted ($p < 0.001$) to reduce the probability of a false positive to less than 5% for the 110 tests, none of the time series fail the diagnostic tests.

a. Significant 1st order autocorrelation in $\ln(R_{t+1}/R_t)$ found.

d. A model with depensatory density-dependence fit the data significantly better than model with no density dependence (Jordan, personal communication; Dennis and Taper, 1994). This indicates that the risk estimates are pessimistic.

t. A model with a trend in m fit the data significantly better than the model with no trend (Jordan, personal communication). This indicates that the risk estimates are optimistic.

l. The variance versus t plot showed significant non-linearity ($R^2 < 0.7$) indicating an underestimate of s^2.

* Trend for Hamilton Springs is positive.

$ i. Index data, no extinction calculation possible; h. no hatchery data, total mixed spawner counts used.

Estimating the variance—The slope method was used to estimate σ^2.

$$\hat{\sigma}^2_{slp} = \text{slope of var}\left[\ln\left(\frac{R_{t+\tau}}{R_t}\right)\right] \quad versus \quad \tau \qquad [8\text{-}4]$$

This method gives estimates of σ^2 that are significantly less biased in the face of severe sampling error (Holmes, in press).

Adjusting parameter estimates for inputs from hatchery-origin spawners—If hatchery fish reproduce successfully in-stream, these inputs must be accounted for, otherwise σ and any risk estimates incorporating σ will be overestimated. This is an accounting problem rather than a negative ecological or genetic effect of the hatchery fish, and arises because σ is qualitatively similar to the number of wild-born offspring divided by the number of parents. If the pool of parents includes both hatchery-origin spawners and wild spawners, the ratio of offspring to parents, and σ, is correspondingly smaller.

Determining the true population-level parameters for the wild component requires both a time series of the proportion of spawners that are of hatchery origin and an estimate of the reproductive success of these spawners. Although hatchery fish appear to have lower breeding success that wild fish (Fleming, 1982; Fleming and Gross, 1993; Fleming and Gross, 1994; Berejikian, 1995), the lifetime reproductive effectiveness of hatchery-origin spawners in the wild has not been well-documented. In the one case where adult-to-adult reproductive success of hatchery fish was compared with that of wild spawners, hatchery fish reproduction was estimated at 10 to 13 percent of that of wild spawners (Chilcote et al., 1986) (however, the hatchery fish in this study originated from non-native broodstock and had been strongly selected for the presence of a genetic marker). Studies comparing survival at specific life stages have found less dramatic differences. For instance, a study measuring survival of hatchery and wild fish from egg to the yearling stage found that in natural settings, the offspring of hatchery parents survived at about 80 percent of the rate that the offspring of wild fish survived (Reisenbichler and McIntyre, 1977).

Given that the information on hatchery fish reproductive success is so sparse and variable, population-level parameters were estimated under two assumptions that taken together, bracket the range of possible situations:

a) Hatchery fish were assumed not to reproduce. That is, all natural spawners observed had wild parents. Parameters were estimated using Equations 8-3 and 8-4 with hatchery spawners removed from the time series before analysis.

b) Hatchery fish were assumed to reproduce at a rate equal to that of wild fish. Wild spawners in the time series may have had wild or hatchery-origin parents. McClure et al. (2000) estimates of μ and σ^2 in this case were (Holmes, 2000):

$$\hat{\mu} = \text{mean}\left[(1-\hat{h}_t)^T + \ln\left(\frac{\sum_{i=1}^{\text{max age}} \varphi_i (S_{w,t-i-1} + eS_{h,t-i-1})}{\sum_{i=1}^{\text{max age}} \varphi_{i-1}(S_{w,t-i-2} + eS_{h,t-i-2})} \right) \right]$$

$$\hat{\sigma}^2 = \text{slope of var}\left[\ln\left(\frac{\sum_{i=1}^{\max i} \varphi_i (S_{w,t-i-1} + eS_{h,t-i-1})}{\sum_{i=1}^{\max i} \varphi_{i-\tau}(S_{w,t-i-1-\tau} + eS_{h,t-i-1-\tau})} \right) \right] \text{versus } \tau \qquad [8\text{-}5]$$

where h_t is the proportion of the spawning population that is of hatchery-origin, S_w is the number of wild spawners, and S_h is the number of hatchery-origin spawners.

(Note that for consistency with the 2000 Federal Columbia River Power System Biological Opinion [NMFS, 2000a], these parameters were also calculated, assuming that hatchery fish reproduce at a rate that is 20 and 80 percent that of wild fish.)

When no estimates of the fraction of hatchery fish were available, parameters were estimated using the total spawner or index count, which may include hatchery fish. In these cases, if the proportion of hatchery fish in the naturally spawning population does not change substantially through time, and those hatchery fish do not reproduce, estimates of μ and σ^2 will be very similar to case 1 (hatchery fish do not reproduce); however, the total population size cannot be estimated since the wild fraction is unknown.

Annual rate of population change—The estimate of the median annual rate of population change (denoted $\hat{\lambda}$ is:

$$\hat{\lambda} = \exp(\hat{\mu}) \qquad [8\text{-}6]$$

Because $\hat{\lambda}$ is distributed lognormally, the median value provides a better indication of the central tendency of the population than the mean. (Note: NMFS uses λ to denote the median while Dennis et al. [1991] use α to indicate the median and λ to indicate the mean.)

The confidence intervals on $\hat{\lambda}$, due to the fact that one uses a finite rather than infinite time series for estimation, were roughly approximated as (Equation 61 in Dennis et al., 1991):

$$\exp(\hat{\mu} - t_{\alpha/2,tq-2}\sqrt{\hat{\sigma}^2/tq}), \exp(\hat{\mu} + t_{\alpha/2,tq-2}\sqrt{\hat{\sigma}^2/tq}) \qquad [8\text{-}7]$$

where $t_{\alpha q}$ is the quantile of a student's-*t* distribution at probability α and degrees of freedom q, and t_q is the length of the running sum time series. In the McClure et al. (2000) application, counts are taken each year, so t_q in the Dennis et al. (1991) equation is simply, $q-1$. This is an overestimate of the true confidence interval since the σ^2 estimate (even using the slope method) contains an upward bias when there is sampling error.

Probability of extinction—For those stocks for which a total live spawner count was available, the risk of absolute extinction (no spawners for an entire generation) was calculated over a 24, 48, and

100 year period. The probability of reaching a particular threshold, in this case $R_e = 1$, from the most recent population size estimate, R_0, within time t_e (Equation 16 x Equation 84 in Dennis et al., 1991) is

$$
\begin{aligned}
G * \pi' = \pi' * \Phi\left(\frac{-\ln(R_0/R_e) + |\hat{\mu}| \ \ t_e}{\hat{\sigma} t_e} \right) \\
+ \exp(2\ln(R_0/R_e)|\hat{\mu}|/\hat{\sigma}^2)\Phi\left(\frac{-\ln(R_0/R_e) - |\hat{\mu}| \ \ t_e}{\hat{\sigma} t_e} \right), \quad t_e > 0
\end{aligned}
\qquad [8\text{-}8]
$$

$$
\text{where} \quad \pi' = \begin{Bmatrix} 1, \hat{\mu} \le 0 \\ \exp(-2\hat{\mu}\ln(R_0/R_e)/\hat{\sigma}^2, \hat{\mu} > 0 \end{Bmatrix}
$$

$$
\text{and } \Phi() \text{ is the inverse normal function.}
$$

The most recent population size estimate, R_0, was set to the most recent running sum estimate:

$$
R_0 = \exp(\hat{\mu}) \sum_{age=1}^{\text{max age}} \varphi_j S_{t-j} \qquad [8\text{-}9]
$$

The R_0 estimate for each stock is given in Table 8-1.

Probability of 90 percent decline—In many cases, the probability of extinction could not be calculated because this probability requires an estimate of the total population size. For instance, total population size estimates were not obtainable if only index counts (such as redds per mile) were available or if no estimate of the fraction of hatchery and wild spawners in the population existed. In four cases, the ESU-level count included only a sample of the total spawners in the ESU, and therefore a population size estimate was not possible. In these cases, the probability that the population is 90 percent lower than its current population size at time t_e (Equation 6 in Dennis et al., 1991) should be relied on:

$$
\Pr\left(\frac{N_{t+\tau}}{N_t} < \frac{10}{1} \right) = \Phi\left(\frac{-\ln(10/1) + \left| \sum_i \mu \right| \ \ t_e}{\sum_i \sigma \ \ t_e} \right) \qquad [8\text{-}10]
$$

Not only is this risk metric amenable to the many situations where a population size estimate was impossible to obtain, it also offers a risk metric that is independent of the initial population size or even the type of count (index or otherwise) and thus can be calculated with greater accuracy. In addition, it estimates an aspect of risk important in populations large enough that extinction is not probable, but the underlying population dynamics are clearly such that the stock is in peril.

8.2.2 Results—Population Trajectories and Risks

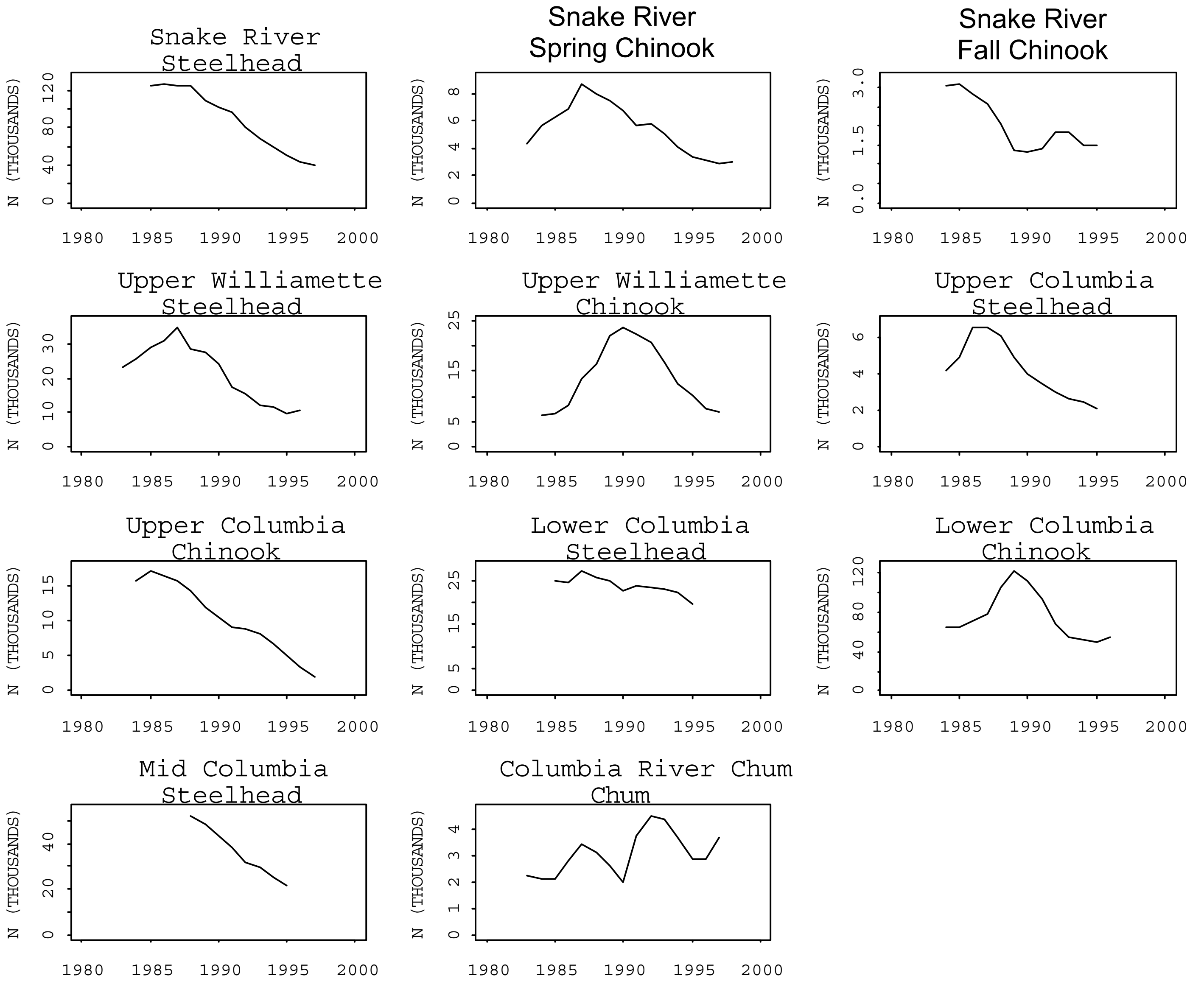
8.2.2.1 Population Trends

Annual population growth rate and the closely related risk of substantial decline have several advantages as measures of population status. Both indicate population status without reference to population size and do not change with different initial abundances. Annual population growth rate also provides a measure by which harvest and other direct impacts on salmon populations can be evaluated. A population with a declining growth rate obviously can not sustain harvest of any form. An increasing population, however, may support such impacts as long as the population growth rate does not fall below one. In fact, managing for lambda may be a reliable means of achieving species viability and productivity because a positive trend (i.e., a lambda value greater than one) will result in more individuals and ultimately a lower extinction risk (Caswell, 2000).

In almost every ESU, the estimated population of actual and potential wild spawners (the weighted running sum) showed marked decline over the time period analyzed (Figure 8-2). Given these trends, it is not surprising that for most stocks and ESUs, the estimated λ was less than one, even in the most optimistic case when it was assumed that hatchery fish had no reproductive output (Figure 8-3 A). In this case, $\hat{\lambda}$ was less than 1.0 for 9 of the 11 ESUs analyzed and less than 0.9 for 2 of these ESUs (Table 8-1). Populations with an annual population growth rate of 0.9 are declining so rapidly that the population can be anticipated to be halved in less than 7 years. At the other extreme, when hatchery fish were assumed to have reproduced at the same rate as wild fish, $\hat{\lambda}$ was correspondingly much lower; all ESUs except Columbia River chum had an estimated λ less than 0.9, and three ESU-level annual population growth rates were less than 0.7 (Table 8-2, Figure 8-3 B). The confidence intervals on λ for several ESUs, particularly Columbia River chum, are large. In general, the high variance in the population growth rate of chum is in part due to the cyclicity this ESU shows over the time period analyzed (Figure 8-2).

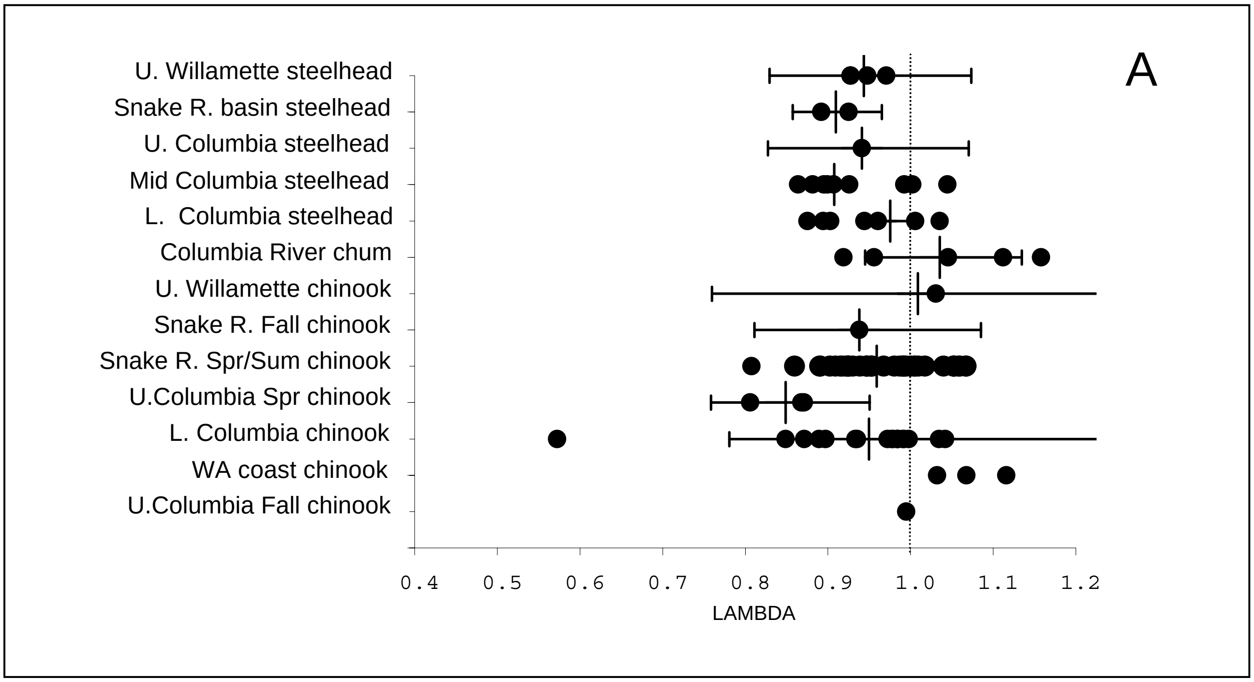
There was greater variation in the estimated λ among stocks than among ESUs (Figure 8-3; Table 8-1). Assuming that hatchery fish do not reproduce, 75 percent of the 95 listed stocks analyzed had an average annual population growth rate less than 1.0, with 20 percent of all stocks having an estimated λ less than 0.9. The estimated population growth rates were increasing or stable for the remaining 25 percent of individual stocks. The Lewis and Clark River chinook (Lower Columbia River ESU), which had 1 or fewer returning spawners over the last 5 years of the data set, had the lowest annual population growth rate ($\hat{\lambda}$ =0.570, confidence interval = 0.21-1.54) and the highest variance (σ^2 = 2.61) of all analyzed stocks. It is worth noting that the Upper Willamette steelhead, Snake River steelhead, and Upper Columbia chinook ESUs did not include a single stock with an increasing or stable trend. The overall pattern is that of severe rates of decline throughout the analyzed stocks; however, when considering the estimates for any specific stock, it should be kept in mind that the confidence intervals for most stocks are large with upper bounds exceeding 1.00. When hatchery fish were assumed to have a reproductive success equal to that of wild fish, $\hat{\lambda}$ was correspondingly much lower (Figure 8-3; Table 8-2). The effects were particularly pronounced in steelhead stocks, which tend to have a high proportion of hatchery-origin fish on the spawning grounds. Again, it is important to note that several stocks and ESUs appear to have an increasing rate of decline through time. Consequently, these estimates of population growth rate, and all risk estimates for these stocks, will be optimistic.

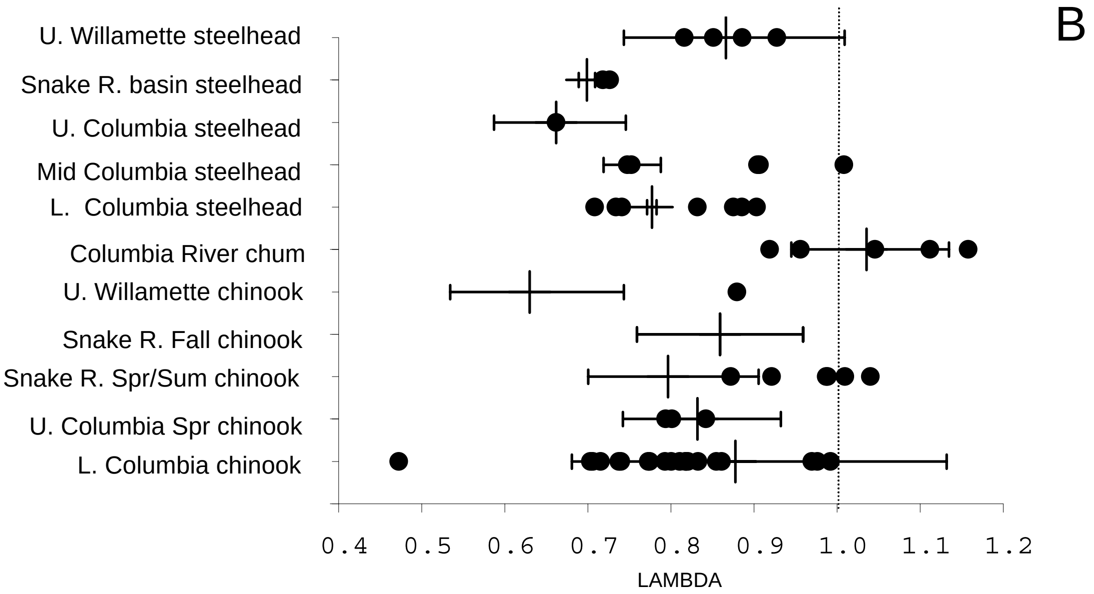
The control stocks widely regarded as healthy had higher estimated rates of annual population growth than the vast majority of threatened or endangered stocks. For example, the growth rates in three Washington coastal chinook stocks ranged from 1.03 to 1.12 (note the lower confidence limit for two of these stocks were below 1.00). The variance for these three stocks was in the lowest



Note: In these plots, R_t was estimated from total (wild + hatchery origin) spawner count time series spanning 1980 to 1999.

Figure 8-2. Time Series of R_t, the Estimated Total Living Current or Future Spawner Population Size, for each ESU in the Columbia River Basin, Plus Hanford Reach and Coastal Chinook





Note: Estimated 95 percent confidence intervals on λ are included for the ESU-level λ estimate. Plot A shows the estimates assuming that no masking of the parameter μ occurred due to hatchery fish reproduction (hatchery reproduction = 0). Plot B shows the estimates assuming that maximal masking of the parameter μ occurred due to hatchery fish reproduction (hatchery reproduction = wild reproduction).

Figure 8-3. Estimated Median Rate of Population Decline, λ, at the Individual Stock Level (black circles) and at the ESU Level (cross-mark)

Table 8-2. Summary of Estimated Population Size (Wild and Hatchery), Parameter Estimates, Risk of Extinction, and 90 Percent Decline in Abundance and Needed Improvements in I to Reduce Risk of Decline or Extinction in 100 Years to Below 5 Percent

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			Population Parameter Estimates					**Risk of Extinction**			**Risk of 90% Decline**			
						95% confidence interval								
ESU	**Stock**	**Pop. size estimate**	**μ**	**σ^2**	**λ**	**low**	**up**	**24 years**	**100 years**	**Req. inc. (%)**	**24 years**	**100 years**	**Req. inc. (%)**	**NA Comments**
Lower Columbia Chinook	**ESU Level**	**NA**	**(0.13)**	**0.14**	**0.88**	**0.68**	**1.13**	**NA**	**NA**	**NA**	**0.67**	**1.00**	**19.0**	**index data**
	Bear Ck	507	(0.35)	0.20	0.71	0.52	0.96	0.87	1.00	46.0	1.00	1.00	50.5	
	Big Ck	5,964	(0.20)	0.04	0.82	0.73	0.92	0.00	1.00	16.0	0.99	1.00	23.5	
	Clatskanie	57	(0.26)	0.44	0.77	0.53	1.14	0.84	1.00	50.5	0.88	1.00	47.0	
	Cowlitz Tule	NA	(0.22)	0.10	0.80	0.66	0.97	NA	NA	NA	0.97	1.00	29.0	index data
	Elochoman	NA	(0.16)	0.43	0.85	0.58	1.25	NA	NA	NA	0.68	0.98	33.0	index data
	Germany	NA	(0.21)	0.14	0.81	0.64	1.02	NA	NA	NA	0.93	1.00	29.0	index data
	Gnat	211	(0.20)	0.45	0.82	0.55	1.21	0.55	0.99	37.0	0.78	1.00	39.5	
	Grays Tule	NA	(0.30)	0.42	0.74	0.51	1.07	NA	NA	NA	0.94	1.00	53.5	index data
	Kalama Spr	NA	(0.30)	0.14	0.74	0.00	0.00	NA	NA	NA	1.00	1.00	41.0	index data
	Kalama	NA	(0.15)	0.52	0.86	0.57	1.31	NA	NA	NA	0.64	0.96	34.5	index data
	Klaskanine	54	(0.26)	0.27	0.77	0.56	1.07	0.87	1.00	42.0	0.93	1.00	40.0	
	Lewis R Bright	NA	(0.03)	0.04	0.97	0.86	1.09	NA	NA	NA	0.06	0.65	4.5	index data
	Lewis Spr	NA	(0.23)	0.42	0.79	0.54	1.15	NA	NA	NA	0.85	1.00	42.5	index data
	Lewis, E Fk Tule	NA	(0.01)	0.02	0.99	0.91	1.08	NA	NA	NA	0.00	0.14	1.0	index data
	Lewis and Clark	1	(0.75)	2.61	0.47	0.18	1.27	1.00	1.00	0.0	0.98	1.00	####	
	Mill Fall	615	(0.35)	0.18	0.70	0.52	0.96	0.87	1.00	45.5	1.00	1.00	50.0	
	Plympton	5,983	(0.18)	0.14	0.83	0.67	1.04	0.01	1.00	18.5	0.87	1.00	25.5	
	Sandy Late	4,263	(0.02)	0.01	0.98	0.89	1.07	0.00	0.00	0.0	0.00	0.53	2.5	
	Skamokawa	NA	(0.33)	0.04	0.72	0.64	0.80	NA	NA	NA	1.00	1.00	41.5	index data
	Youngs	38	(0.20)	1.04	0.82	0.42	1.58	0.78	0.99	81.0	0.69	0.96	56.5	
Upper Columbia spring chinook	**ESU Level**	**2,152**	**(0.18)**	**0.04**	**0.83**	**0.74**	**0.93**	**0.00**	**1.00**	**15.5**	**0.99**	**1.00**	**21.5**	
	Methow	433	(0.17)	0.21	0.84	0.65	1.09	0.25	1.00	23.5	0.79	1.00	26.5	
	Entiat	173	(0.22)	0.04	0.80	0.72	0.90	0.60	1.00	23.5	1.00	1.00	26.5	
	Wenatchee	805	(0.23)	0.03	0.79	0.72	0.87	0.08	1.00	21.5	1.00	1.00	26.5	

Table 8-2. Summary of Estimated Population Size (Wild and Hatchery), Parameter Estimates, Risk of Extinction and 90 Percent Decline in Abundance and Needed Improvements in I to Reduce Risk of Decline or Extinction in 100 Years to Below 5 Percent

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		Population Parameter Estimates						Risk of Extinction			Risk of 90% Decline			
ESU	Stock	Pop. size estimate	μ	σ^2	λ	95% confidence interval low	up	24 years	100 years	Req. inc. (%)	24 years	100 years	Req. inc. (%)	NA Comments
Snake River spring/summer chinook	**ESU Level**	**72,497**	**(0.23)**	**0.06**	**0.80**	**0.70**	**0.91**	**0.00**	**1.00**	**17.5**	**1.00**	**1.00**	**28.0**	
	Bear Ck	736	0.02	0.15	1.02	0.83	1.25	0.00	0.03	0.0	0.07	0.15	3.0	
	Imnaha R	1,175	(0.14)	0.03	0.87	0.79	0.96	0.00	1.00	10.5	0.88	1.00	15.5	
	Johnson Ck	457	0.01	0.05	1.01	0.00	0.00	0.00	0.00	0.0	0.01	0.07	0.5	
	Marsh Ck	291	(0.01)	0.13	0.99	0.82	1.19	0.00	0.19	3.0	0.13	0.39	5.5	
	Minam R	582	(0.08)	0.17	0.92	0.74	1.15	0.02	0.77	11.0	0.43	0.93	14.5	
	Poverty Ck	1,055	(0.01)	0.10	0.99	0.84	1.17	0.00	0.05	0.5	0.09	0.35	4.5	
	Sulphur Ck	207	0.04	0.41	1.04	0.74	1.47	0.05	0.21	7.0	0.15	0.17	8.5	
	Alturas Lake Creek	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	index data; no hatchery data
	American River	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	index data; no hatchery data
	Bear Valley Creek	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	index data; no hatchery data
	Big Sheep Creek	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	index data; no hatchery data
	Bushy Fork	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	index data; no hatchery data
	Catherine Creek	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	index data; no hatchery data
	Crooked Fork	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	index data; no hatchery data
	Elk Creek	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	index data; no hatchery data
	Grande Ronde River	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	index data; no hatchery data
	Lemhi River	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	index data; no hatchery data
	Loon Creek	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	index data; no hatchery data
	Lostine Creek	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	index data; no hatchery data
	Lower Salmon River	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	index data; no hatchery data
	Lower Valley Creek	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	index data; no hatchery data
	Moose Creek	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	index data; no hatchery data
	Newsome Creek	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	index data; no hatchery data
	Red River	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	index data; no hatchery data
	Salmon River E Fk	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	index data; no hatchery data
	Salmon River S Fk	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	index data; no hatchery data
	Secesh River	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	index data; no hatchery data
	Selway River	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	index data; no hatchery data

Table 8-2. Summary of Estimated Population Size (Wild and Hatchery), Parameter Estimates, Risk of Extinction and 90 Percent Decline in Abundance and Needed Improvements in I to Reduce Risk of Decline or Extinction in 100 Years to Below 5 Percent

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		Population Parameter Estimates						Risk of Extinction			Risk of 90% Decline			
ESU	Stock	Pop. size estimate	μ	σ^2	λ	95% confidence interval low	up	24 years	100 years	Req. inc. (%)	24 years	100 years	Req. inc. (%)	NA Comments
	Upper Big Creek	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	index data; no hatchery data
	Upper Salmon River	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	index data; no hatchery data
	Upper Valley Creek	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	index data; no hatchery data
	Wallowa Creek	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	index data; no hatchery data
	Wenaha River	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	index data; no hatchery data
	Whitecap Creek	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	index data; no hatchery data
	Yankee Fork	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	index data; no hatchery data
	Yankee West Fork	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	index data; no hatchery data
Snake River Basin chinook	**ESU Level**	**2,199**	**(0.15)**	**0.01**	**0.86**	**0.00**	**0.00**	**0.00**	**1.00**	**10.0**	**0.99**	**1.00**	**16.0**	
	Snake River Basin Fall	2,199	(0.15)	0.01	0.86	0.80	0.92	0.00	1.00	10.0	0.99	1.00	16.0	
Upper Willamette chinook	**ESU Level**	**44,666**	**(0.46)**	**0.08**	**0.63**	**0.53**	**0.74**	**0.63**	**1.00**	**50.5**	**1.00**	**1.00**	**62.5**	
	McKenzie R.	6,859	(0.13)	0.24	0.88	0.66	1.17	0.01	0.85	15.0	0.63	0.98	22.0	
Upper Columbia fall chinook	Hanford Reach	NA	NA	NA	NA	0.85	1.17	NA	NA	NA	NA	NA	NA	h
WA Coast chinook	Hoh R Fall	NA	NA	NA	NA	0.95	1.13	NA	NA	NA	NA	NA	NA	h
	Queets R Fall	NA	NA	NA	NA	1.01	1.23	NA	NA	NA	NA	NA	NA	h
	Willapa R Fall	NA	NA	NA	NA	0.98	1.16	NA	NA	NA	NA	NA	NA	h
Columbia River chum	**ESU Level**	**NA**	**0.03**	**0.03**	**1.04**	**0.94**	**1.13**	**NA**	**NA**	**NA**	**0.00**	**0.00**	**0.0**	**index data**
	Grays R west fk	NA	0.21	0.20	1.23	0.00	0.00	NA	NA	NA	0.00	0.00	0.0	index data
	Grays R	NA	(0.04)	0.12	0.96	0.78	1.16	NA	NA	NA	0.24	0.73	8.5	index data
	Hardy Ck	NA	0.04	0.06	1.05	0.92	1.19	NA	NA	NA	0.00	0.00	0.0	index data
	Crazy J	NA	0.15	0.03	1.16	1.05	1.28	NA	NA	NA	0.00	0.00	0.0	index data
	Hamilton	NA	(0.08)	0.05	0.92	0.81	1.05	NA	NA	NA	0.40	1.00	10.5	index data
	Hamilton Sprs	NA	0.11	0.59	1.11	0.74	1.68	NA	NA	NA	0.10	0.10	6.0	index data

Table 8-2. Summary of Estimated Population Size (Wild and Hatchery), Parameter Estimates, Risk of Extinction and 90 Percent Decline in Abundance and Needed Improvements in I to Reduce Risk of Decline or Extinction in 100 Years to Below 5 Percent Page 4 of 5

		Population Parameter Estimates						Risk of Extinction			Risk of 90% Decline			
ESU	Stock	Pop. size estimate	μ	σ^2	λ	95% confidence interval low	up	24 years	100 years	Req. inc. (%)	24 years	100 years	Req. inc. (%)	NA Comments
Lower Columbia steelhead	**ESU Level**	**NA**	**(0.25)**	**0.00**	**0.78**	**0.77**	**0.78**	**NA**	**NA**	**NA**	**1.00**	**1.00**	**26.0**	**index data**
	Clackamas Sum	9,065	(0.34)	0.01	0.71	0.66	0.76	0.05	1.00	31.5	1.00	1.00	40.5	
	Clackamas Win	3,123	(0.31)	0.00	0.73	0.71	0.76	0.02	1.00	27.5	1.00	1.00	35.0	
	Green R Win	660	(0.10)	0.21	0.90	0.58	1.41	0.06	0.86	14.0	0.53	0.96	18.0	
	Kalama R Sum	18,843	(0.30)	0.02	0.74	0.68	0.80	0.00	1.00	25.0	1.00	1.00	35.0	
	Kalama R Win	6,294	(0.12)	0.01	0.89	0.84	0.94	0.00	1.00	5.5	0.93	1.00	12.0	
	Sandy Win	6,012	(0.18)	0.03	0.83	0.75	0.92	0.00	1.00	14.0	0.99	1.00	21.0	
	Toutle Win	3,008	(0.13)	0.00	0.88	0.86	0.89	0.00	1.00	6.0	1.00	1.00	12.5	
Mid Columbia steelhead	**ESU Level**	**NA**	**(0.28)**	**0.00**	**0.75**	**0.72**	**0.79**	**NA**	**NA**	**NA**	**1.00**	**1.00**	**31.5**	**index data**
	Deschutes R Sum	70,500	(0.29)	0.02	0.75	0.68	0.82	0.00	1.00	22.5	1.00	1.00	34.0	
	Mill Ck Sum	NA	NA	NA	NA	0.84	1.17	NA	NA	NA	NA	NA	NA	No hatchery data
	Shitike Ck Sum	NA	NA	NA	NA	0.88	0.98	NA	NA	NA	NA	NA	NA	No hatchery data
	Warm Spr Nfh Sum	1031	(0.10)	0.05	0.91	0.76	1.08	0.00	0.92	7.5	0.52	1.00	12.0	
	Eightmile Ck Win	NA	NA	NA	NA	0.00	0.00	NA	NA	NA	NA	NA	NA	No hatchery data
	Ramsey Ck Win	NA	NA	NA	NA	0.59	1.71	NA	NA	NA	NA	NA	NA	No hatchery data
	Fifteen Mile Ck Win	NA	NA	NA	NA	0.77	1.07	NA	NA	NA	NA	NA	NA	No hatchery data
	Umtilla R Sum	9,809	(0.10)	0.00	0.90	0.85	0.96	0.00	0.91	2.5	0.64	1.00	9.5	
	Yakima R Sum	5,561	0.01	0.01	1.01	0.92	1.10	0.00	0.00	0.0	0.00	0.00	0.0	
Upper Columbia steelhead	**ESU Level**	**7,708**	**(0.41)**	**0.03**	**0.66**	**0.59**	**0.75**	**0.87**	**1.00**	**43.0**	**1.00**	**1.00**	**52.5**	
	Upper Columbia R	7,708	(0.41)	0.03	0.66	0.59	0.75	0.87	1.00	43.0	1.00	1.00	52.5	
Snake River Basin steelhead	**ESU Level**	**379,578**	**(0.36)**	**0.00**	**0.70**	**0.69**	**0.71**	**0.00**	**1.00**	**26.5**	**1.00**	**1.00**	**40.5**	
	Snake R A-run	299,161	(0.33)	0.00	0.72	0.71	0.73	0.00	1.00	23.5	1.00	1.00	37.0	
	Snake R B-run	100,455	(0.32)	0.02	0.73	0.00	0.00	0.00	1.00	26.0	1.00	1.00	38.0	

Table 8-2. Summary of Estimated Population Size (Wild and Hatchery), Parameter Estimates, Risk of Extinction and 90 Percent Decline in Abundance and Needed Improvements in I to Reduce Risk of Decline or Extinction in 100 Years to Below 5 Percent

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			Population Parameter Estimates					Risk of Extinction			Risk of 90% Decline			
ESU	Stock	Pop. size estimate	μ	σ^2	λ	95% confidence interval low	95% confidence interval up	24 years	100 years	Req. inc. (%)	24 years	100 years	Req. inc. (%)	NA Comments
Upper Willamette steelhead	**ESU Level**	**12,443**	**(0.14)**	**0.07**	**0.87**	**0.74**	**1.01**	**0.00**	**0.98**	**10.5**	**0.81**	**1.00**	**18.0**	
	Mollala	2,644	(0.20)	0.11	0.82	0.67	0.99	0.04	1.00	21.0	0.94	1.00	26.5	
	N Santiam R	5,653	(0.12)	0.06	0.89	0.77	1.02	0.00	0.94	8.5	0.70	1.00	15.0	
	S Santiam	3,730	(0.16)	0.06	0.85	0.74	0.98	0.00	1.00	13.0	0.91	1.00	19.5	
	Calapooia	416	(0.08)	0.19	0.93	0.72	1.19	0.04	0.74	11.0	0.41	0.88	14.0	

Note: These estimates assume that hatchery fish on the spawning grounds reproduce at a rate equal to that of wild fish. This analysis requires an estimate of hatchery fraction in the spawner count; stocks with no hatchery fraction estimate (see Table 8-1) are not included. Estimates are provided for individual stocks and ESUs (in bold).

15 percent of all stocks analyzed. The population growth rate of the Hanford Reach fall chinook was slightly below 1.00 ($\hat{\lambda}$ = 0.995; confidence interval = 0.85-1.17), reflecting that on average, this population has been in decline over the last 20 years, and highly variable during that time.

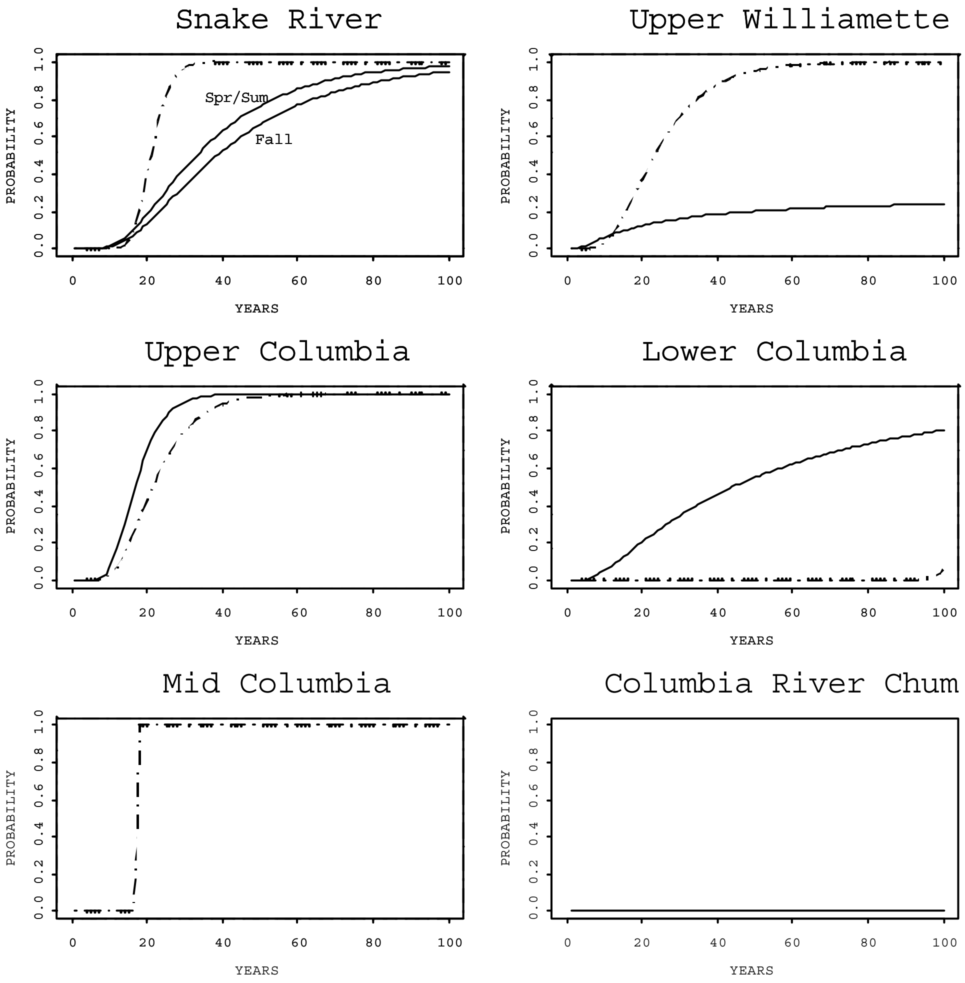
8.2.2.2 Comparative Risks

If conditions and population patterns prevalent from 1980 to the present continue, both individual stocks and ESUs as a whole are at substantial risk of severe declines. When hatchery fish were assumed to contribute nothing to subsequent generations, (which gave the highest estimates of σ), the short-term (24 years) risk of 90 percent declines was equal to or greater than 20 percent for four of the ESUs (Table 8-1). The risk increased with time, and in the long term (100 years), the calculated risk of such a decline was virtually certain (> 90 percent) for 8 ESUs (Table 8-1, Figure 8-4). If hatchery and wild fish reproduce at the same rate, the estimates of in-stream reproduction are much lower and the risk of 90 percent decline correspondingly much higher. In this case, all ESUs except Columbia River chum had a very high probability (> 67 percent) of declining by 90 percent in 24 years or less (Table 8-2). Recall that this is not a negative effect of hatchery fish, but an estimation effect. The parent pool is larger (and the offspring/parents ratio lower) if reproductive hatchery fish are continually input into the system. (Results assuming that hatchery fish contribute to future generations at a rate 20 percent and 80 percent of that of wild fish are shown in Annex D. These numbers are consistent with those used in the 2000 FCRPS Biological Opinion [NMFS, 2000a].)

At the stock level, there was great variation in the probability of 90 percent decline (Figure 8-4). However, the risk of declining in abundance by 90 percent in the long-term (100 years) was greater than 50 percent for nearly two-thirds of the stocks evaluated, even under the most optimistic scenario (no hatchery fish reproduction). When hatchery fish were assumed to reproduce, the risks were correspondingly higher: 45 out of 56 stocks had a 50 percent or greater chance of these serious declines in the long-term (Tables 8-1 and 8-2). Unfortunately, in many cases these did not appear to be gradual declines. Nearly 30 percent of stocks had a greater than 50 percent chance of realizing these substantial declines in 24 years or less, even when the population trajectories were not masked by hatchery fish (i.e., hatchery fish do not reproduce). Risk of decline tended to be higher for steelhead, which generally had lower estimated annual population growth rates over the 1980 to present time period.

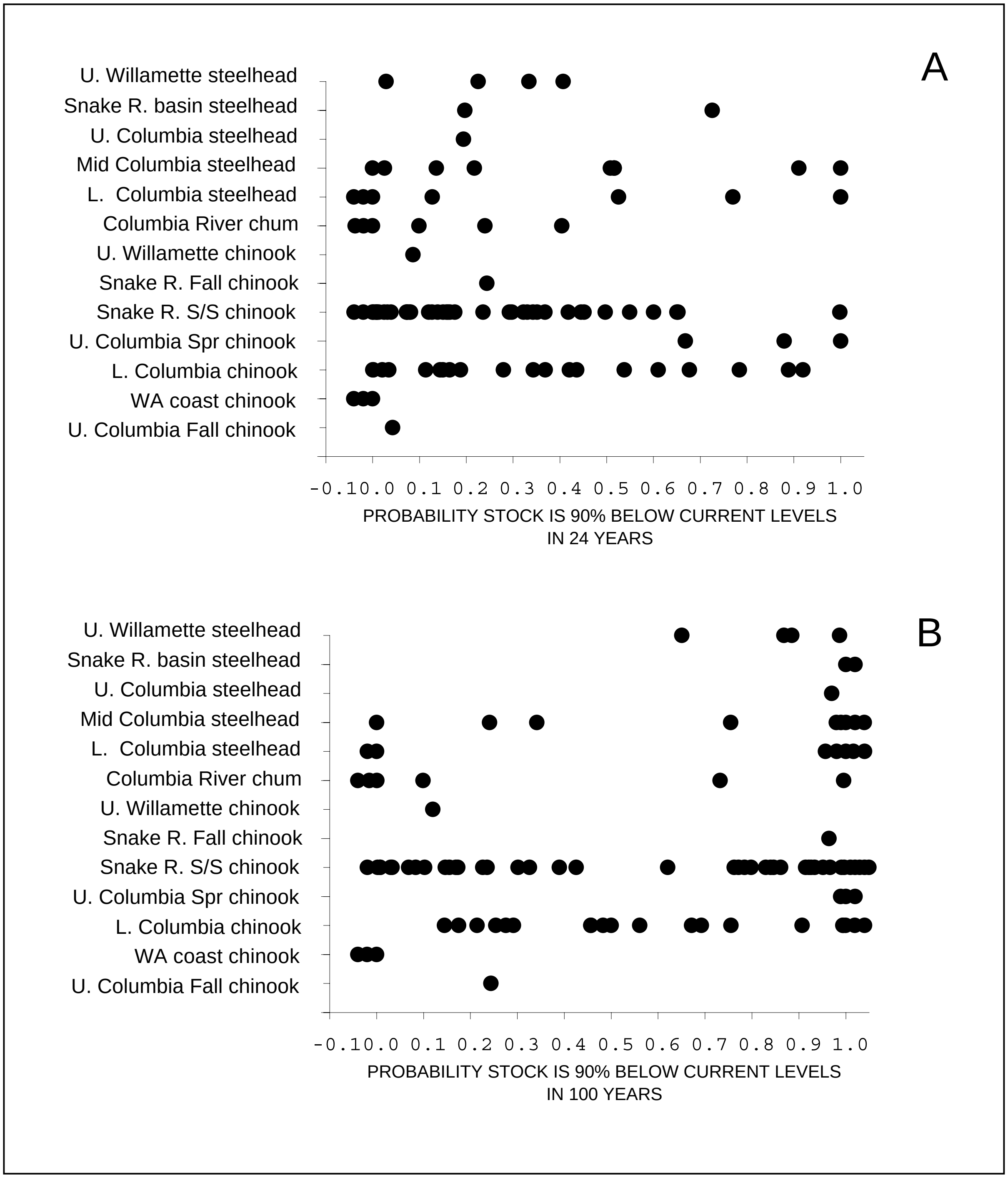
The probability of extinction, calculated for the 40 stocks for which an estimate of the total current population size was possible, indicated that, as expected, risk varies with the time frame analyzed (Figure 8-5; Table 8-1). In the short-term, extinction risks were relatively low for all stocks, even under the most pessimistic parameter estimates (when high hatchery fish reproduction is assumed). However, in the long-term, extinction risks were substantial. With parameters estimated assuming no hatchery fish masking (i.e., hatchery fish reproduction equals zero), half of the stocks had a greater than 50 percent chance of absolute extinction, and 20 percent had an extinction probability of 1.00. These numbers increase to 82 and 62 percent respectively, if parameter estimates assumed high hatchery fish masking (i.e., hatchery fish reproduction equals that of wild fish) (Table 8-2). Again, steelhead stocks tended to be at slightly greater risk than chinook stocks.

Several mathematical biologists have recently cautioned that point estimates of extinction risk typically have such large confidence intervals that the estimates become meaningless (Ludwig,



Note: For this plot, parameter estimates were made assuming that no masking of the parameter μ occurred due to hatchery fish reproduction (hatchery reproduction = 0). When multiple points overlap at 0 or 1, the numbers have been adjusted up or down to make the overlaying points visible.

Figure 8-4. Estimated Probability of that and ESU is 90 Percent Below Current Levels at a Given Number of Years in the Future



Note: Results are shown at the individual stock level (black circles) and at the ESU level (cross-mark). The parameters were estimated assuming that no masking of the parameter μ occurred due to hatchery fish reproduction (hatchery reproduction = 0). When multiple points overlap at 0 or 1, the numbers have been adjusted up or down to make the overlaying points visible.

Figure 8-5. Estimated Probability that a Stock (or ESU) is 90 Percent Lower at 24 Years (Plot A) or 100 Years (Plot B)

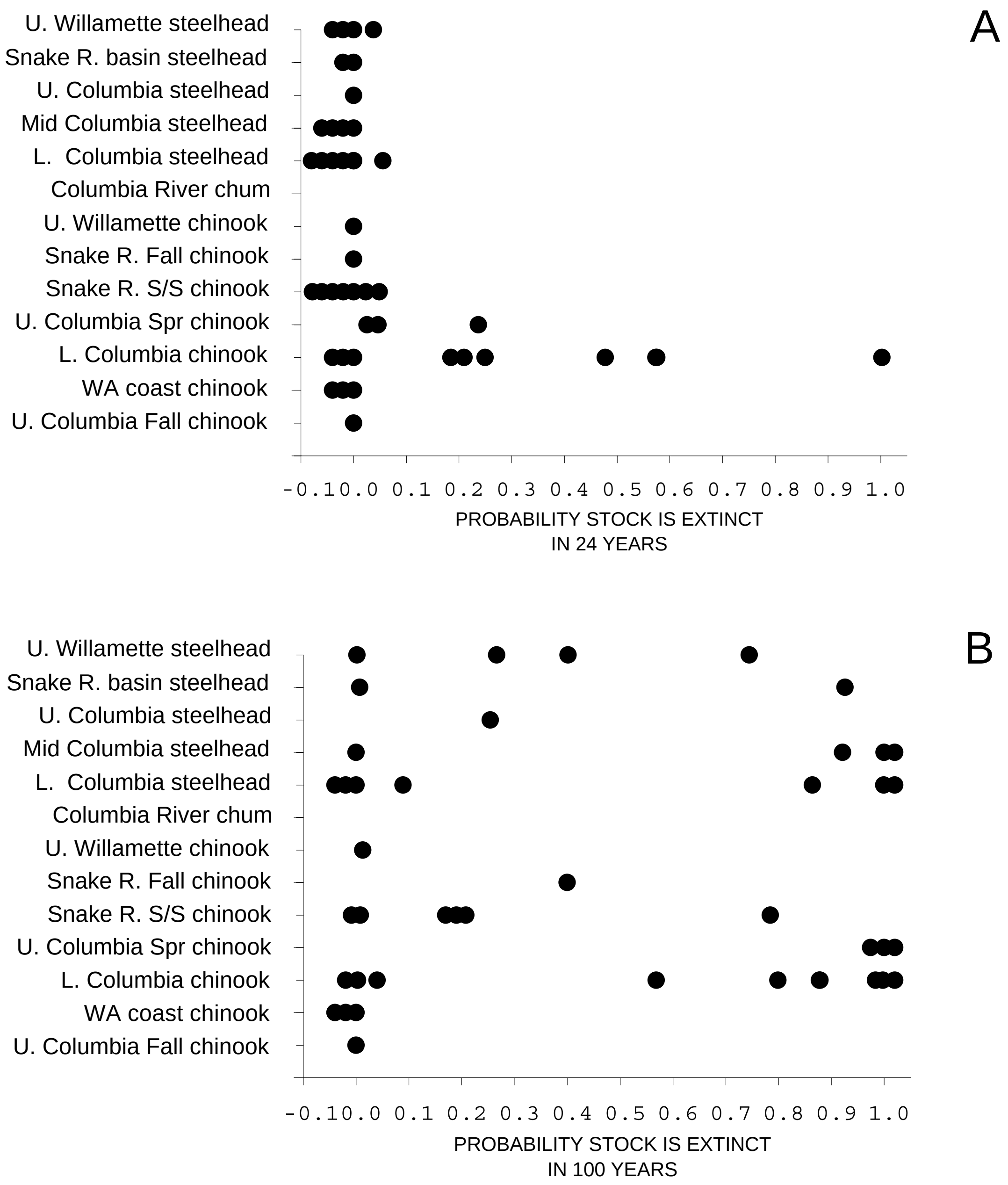
1999; Fieberg and Ellner, 2000). However, other evaluations suggest that simple viability models, which are over-simplifications, still perform surprisingly well (Fagan et al., 1999; Brook et al., 2000; Caswell, 2000; Meir and Fagan, 2000). The extinction risk estimates calculated by McClure et al. (2000) are most properly viewed as measures of relative risk rather than absolute predictions because they do not incorporate genetic or demographic effects operating at low abundance levels, catastrophic events, or any number of other factors known to increase or decrease the risk of extinction. However, they are the only metric McClure et al. (2000) use that incorporates current population size as well as trend and variance.

Not surprisingly, risks faced by the healthy stocks were generally at the positive extreme of the distribution. The risk of extinction was zero for all four healthy control stocks, in both the short and the long-term. Washington Coastal chinook stocks also had no risk of a 90 percent decline in either time frame. The Hanford Reach fall chinook, however, did have a 24 percent chance of experiencing a 90 percent decline in abundance within 100 years (Table 8-1). In fact, this stock has already experienced a 90 percent drop in abundance from its peak in the mid-1980s.

Correspondence between the different risk metrics was relatively good (Figure 8-6). The correlation between long-term risk of extinction and decline was high (Pearson $r = 0.731$, $p = 0.000$) although the values of the two metrics did not always align perfectly. A few stocks, most notably Youngs River chinook in the Lower Columbia ESU, had lower risks of decline than extinction. In these cases, the λ estimate was close to one, but current population size was low and the estimate of variance was large. Small population size and high variability increases the vulnerability of these stocks to extinction, even though the mean population trajectory is only slightly less than one. More often, stocks had a relatively low risk of extinction, but a high risk of decline in the long-term. This situation is illustrated in the extreme by the Clackamas winter steelhead (in the Lower Columbia ESU) and the Snake River steelhead A-run, both of which had a long-term extinction risk at or near zero, but a 100 percent risk of realizing a substantive decline. In both cases, the annual population growth rate was below one, and the variance in the instantaneous rate of change was very low, indicating that these stocks are following a relatively straight downward trajectory. However, the current population size of both stocks is large enough that the risk of going extinct, even in a 100-year time frame, is relatively low. Clearly, neither measure of risk (decline or extinction) fully captures population status, and considering both types of risk in management and conservation efforts will be important.

8.2.2.3 Needed Changes to Mitigate Extinction Risk/Risk of Decline

Because the ultimate goal of conservation efforts is improving the status of imperiled species or populations, estimates of annual population growth and risk were used to determine how much change in population trajectories is necessary to mitigate the current risks. At both stock and ESU levels, McClure et al. (in review) calculated the percent increase in λ necessary to reduce the probability of 90 percent decline in 100 years to less than 5 percent. In addition, when estimates of total population size were available, they calculated the percent increase in λ necessary to reduce the risk of extinction to less than 5 percent in 100 years. Although these calculations do not suggest specific management actions necessary to increase population growth rates, they do contribute to establishing management goals. The potential for changes in variance to reduce risks of decline or extinction for these stocks was not evaluated, although this does present another way in which



Note: Black circles indicate those individual stocks for which a population size estimate was possible (total live spawner count and hatchery fraction available). The parameters were estimated assuming that no masking of the parameter μ occurred due to hatchery fish reproduction (hatchery reproduction = 0).

Figure 8-6. Estimated Probability of Extinction within 24 Years (Plot A) or 100 Years (Plot B) management actions might alter the status of the stocks. Reducing the variance in the rate of instantaneous change would reduce the risk of a population going extinct merely by stochastic processes, for instance. Alternatively, if populations are dependent on high recruitment years for viability, reducing this variance may in itself endanger the populations.

In most cases (except where the variance is very high), the change needed to reduce the risk of decline is greater than that required to reduce the risk of extinction. To reduce the risk of a 90 percent decline in a 100-year time period, necessary improvements in annual population growth rate at the stock level ranged from 0 to 65 percent, with a mean of 9 percent (Table 8-1). Reducing the long-term risk of extinction required improvements ranging from 0 to 174 percent (mean, 12 percent) (Table 8-1). The greater improvements required to avoid long-term declines are due in part to the fact that large, slightly declining populations can avoid reaching the extinction threshold over the analyzed time frame. Chinook stocks generally required greater improvements in λ to mitigate risk than did steelhead stocks, in spite of the slightly lower λ and higher risk of substantial decline for steelhead stocks. This is due to the interaction between the rate of instantaneous increase, σ, and the variance in that rate. Individual chinook stocks tended to be small with high variability, whereas steelhead stocks were generally larger (sometimes entire basins) with lower variability. As with other measures, if hatchery fish contribute to subsequent generations, the current population growth rate is lower, and the needed improvements are larger (Table 8-2).

8.2.3 Implications for Columbia River Basin Conservation Planning

Regardless of the risk metric chosen, the 12 listed salmonid ESUs in the Columbia River Basin are clearly imperiled. Even under the most optimistic assumptions, 9 of 11 ESUs had declining annual population growth rates. One (Snake River sockeye) is currently so low in abundance as to be virtually extinct. In the remaining two ESUs, which had positive estimated population trends, the lower confidence intervals around the population growth rate estimates extended well below one. Nineteen of 40 stocks had a calculated extinction risk equal to or greater than 50 percent in 100 years; 59 percent of all stocks face a greater than 50 percent chance of a 90 percent decline in abundance in that time. Even in a 24-year period, nearly one-third of the stocks analyzed had a greater than 50 percent chance of this serious decline. If the parameter estimates were adjusted for hatchery fish reproduction, the situation would look even more bleak, particularly for steelhead stocks.

In contrast to the threatened and endangered ESUs in the Columbia River Basin are the three Washington Coastal chinook stocks widely regarded as “healthy.” These three stocks all had estimated population growth rates greater than one, and no risk of extinction or serious decline in the short- or long-term. Demographically, at least, these populations appear to be viable. Thus, the demographic parameters characterizing these stocks, annual population growth rate, variance, and population size, provide useful points of comparison for assessing the status of the listed stocks.

More problematic, however, was the Hanford Reach fall chinook stock. This population exhibited a dramatic peak in abundance in the mid-1980s, and has been declining since that time. This pattern yielded an annual population growth rate very slightly less than one ($\lambda = 0.995$), with a very wide confidence interval. Although this stock had no risk of extinction, due to its extremely large population size, it did have some chance (24 percent) of sustaining a serious decline in the long-term. The most conservative interpretation of these results is that if current conditions continue, including the very high harvest rates prevalent throughout the analyzed time period, this “healthy” stock stands a considerable chance of experiencing a long-term decline. A less pessimistic reading of these results might suggest that this population exhibits large cycles in abundance, and may rebound naturally. The ambiguity in the status of this very large stock underscores the need for continued monitoring of even apparently vigorous populations.

These results also underscored the importance of considering population structure in viability analyses. Salmon data have been traditionally collected on a stream-by-stream basis and treated as separate populations. However, fish in multiple streams or rivers may belong to a single population. Geographically based “population” parameters may be misleading in these cases; similarly, one or more populations may serve as a source for other sink populations. For example, the Lewis and Clark River chinook stock began the analyzed time series with approximately 100 returning spawners but had no returning spawners for the last 5 years of the available time series. However, in the years prior to 1980, this stream had no returning spawners for several years in the early 1970s and fewer than 10 returning spawners for several years in the 1950s. This river may be a local sink, experiencing repeated local extinctions and re-colonizations (Pulliam, 1988; Pulliam and Danielson, 1991). In fact, both the Lower Columbia chinook and Snake River spring/summer chinook ESUs contained several stocks with population growth rates above one, as well as a majority of stocks with declining population trends, suggesting that several “good” populations may be supporting other weaker populations.

The possibility that populations have not been appropriately defined or that source-sink dynamics are present in Columbia River salmonids raises two important issues. First, because recovery planning efforts depend on estimates of the status of populations, it is critical that those populations be biologically (rather than geographically) defined. Estimates of dispersal rates among stocks will be an important component of determining population boundaries, and will ultimately provide the most appropriate risk assessments in support of recovery planning (McElhany et al., 2000). Second, the presence of adults from a source population has the potential to complicate the interpretation of adult census data (Brawn and Robinson, 1996), much as the presence of hatchery fish on the spawning grounds can complicate census data. Determining age-specific survival rates, which can provide a more robust picture of the status of populations in a specific area, should thus be an important complement to collecting census data. In sum, defining populations, including estimates of dispersal rates, and acquiring more detailed demographic information are critical components of good recovery planning.

8.2.4 A Note on Density Independence and Risks to Populations

The Ricker function and its many modifications have a long history as the premier population growth models employed in fisheries biology. The Ricker model assumes that the log of the rate of recruitment per spawner decreases linearly as spawner density increases, and it is the model underlying all PATH simulations for Snake River chinook salmon. A critically important parameter for assessing extinction risk is the per capita production of recruits when populations are low (near extinction), which can be estimated from a Ricker model as the intercept of the linear regression relating natural log of “recruits per spawner” to the number of spawners (this is the “A” referred to in the PATH models for Snake River chinook salmon). In practice, estimates of this parameter based on a Ricker function are biased toward producing unduly optimistic portraits of the future for populations (Ginzburg et al., 1990), because they assume that there will be greater recruitment as the number of spawners decreases.

It is worth noting that to date, most extinction risk analyses applied to salmonid populations have relied upon density-dependent models. For example, Emlen (1995) fit Ricker equations to counts of chinook salmon redds (nests) from 1957 to 1992 and used the estimated productivity at low density (or '□-value') as a parameter in a stochastic model of population growth. Emlen concluded that:

> *the… present estimated α-value apparently is sufficient to virtually ensure population persistence over the next 100 years, and to lead to considerable increases in the number of redds over present counts… Population recovery, also, might be expected under present α. Indeed, in the absence of adverse weather conditions, environmental deterioration, or unexpected setbacks, the 1957-1961 levels should be regained within about 100 years." (Page 1,447.)*

In contrast to these predictions, redd counts have continued to decline in these same streams (data for 1993 through 1995), and several of the populations are perilously close to extinction. For example, the 1995 summed redd count for Bear Valley and Elk was only 8 redds, whereas the summed count for the same areas historically hovered around 1,000 redds. Ratner et al. (1997) similarly incorporated density dependence in their stochastic population projections of chinook salmon in Oregon. Using a Ricker function to estimate the probability of survival from eggs to smolts, they concluded that "under the assumption of no further habitat destruction, the population is predicted to have a greater than 95 percent probability of persistence for 200 years."

Schaller et al. (1999) found density dependence in Snake River spring/summer chinook stocks. However, Schaller et al.(1999) detect a strong density signal only when data spanning from 1939 until 1990 are used and when they combine all index stocks into a single aggregate population. As stated in McClure et al. (2000) and Kareiva et al. (2000), compensatory density dependence does not appear over the 1980 to 1999 time period. Therefore, for the purpose of extinction analyses aimed at assessing the risk of losing particular stocks, NMFS feels it is better to treat each index stock separately, and to examine the data from 1980 onward as representative of current conditions. If populations rebuild to the very high levels seen prior to 1970, then density-independent analyses would be grossly in error (however, if this were the case, the populations would have recovered), and there would no longer be a need for an extinction risk analysis. The apparent discrepancy between PATH and CRI analyses with regard to density dependence may also be due, in part, to different definitions of recruits. CRI tabulates recruits at the spawning ground, whereas PATH adds losses due to harvest and upstream mortality, and tabulates recruits at the mouth of the Columbia River. The CRI data are closer to the actual observations, because they do not require back calculations involving estimates of upstream losses.

8.3 Using Matrix Models to Summarize Demographic Rates and Explore Opportunities for Recovery

The preceding analyses provide estimates of population growth rate and associated risks. The next steps are to explore what is known about the life cycle of particular stocks and describe where mortality occurs; both steps are needed to identify opportunities for recovery. Demographic matrices are mathematical devices for organizing schedules of mortality and reproduction into a framework convenient for data presentation, analysis, and prediction. Year-class matrices have been adopted to iterate salmonid populations from one year to the next, as shown in the following example:

$$N(t+1) = A * N(t) \qquad [8\text{-}11]$$

where N(t) is a column vector pertaining to the number of individuals in each of the five age classes:

$$\begin{array}{l} N_1 \\ N_2 \\ N_3 \\ N_4 \\ N_5 \end{array}$$

with N_x corresponding to number of fish of age x. The matrix A is a 5-by-5 matrix with the following structure:

$$A = \begin{array}{ccccc} 0 & 0 & R_3 & R_4 & R_5 \\ a_{12} & 0 & 0 & 0 & 0 \\ 0 & a_{23} & 0 & 0 & 0 \\ 0 & 0 & a_{34} & 0 & 0 \\ 0 & 0 & 0 & a_{45} & 0 \end{array} \qquad [8\text{-}12]$$

where the above matrix would pertain to fish that live, at most, 5 years, but that could reproduce as early as year 3. The top row represents production of young from 3-, 4-, or 5-year-old fish, and the a_{ij} along the sub-diagonal represents transitions of fish from the *i*th age class to the *j*th age class. Each element in the matrix may actually be more complicated than displayed above. For example,

$$R_3 = (1-s)b_3(m_3/2)s_1 \qquad [8\text{-}13]$$

where s is the mortality of adult females as they swim upstream to spawn, b_3 is the propensity of 3-year-old females to migrate upstream to breed, m_3 is the fecundity of age 3 females, and s_1 is the survival from eggs to 1-year-olds. Similarly, instead of a simple a_{ij} transition rate for survival from one age class to the next, complications must be accounted for. For instance, when modeling the fate of fish from Snake River stocks between their first and second birthday (a_{12}), the fact that fish may experience different survival rates depending on whether they are barged down the river or swim down the river must be recognized. Thus, a_{12} for Snake River stocks may be expressed as:

$$a_{12} = ((1-pt) * s_d + pt*s_b)* s_e \qquad [8\text{-}14]$$

where pt is the proportion of fish transported in barges, s_d is survival of fish that swim downstream, s_b is survival of barged fish, and s_e is survival of smolts in the estuary and during their first winter in the ocean. Equation 8-14 neglects the hypothesis favored by some biologists that survival in the estuary and early ocean phases depends on whether fish were barged or swam to the estuary, but it would be easy to expand s_e into two separate terms that parameterize this hypothesized complication. For the older age classes, the a_{ij} is more straightforward:

$$\begin{array}{l} a_{23} = s_3 \\ a_{34} = s_4(1-b_3) \\ a_{45} = s_5(1-b_4) \end{array} \qquad [8\text{-}15]$$

where s_x is the survival from age x-1 to age x, and b_x is the propensity of adults of age x to breed.

This basic matrix framework is exceptionally flexible and can accommodate:

- density dependence in particular matrix elements
- dispersal between different populations
- life history variation, with transitions from one life history to another
- impacts of all four "H" factors
- environmental variability and uncertainty in parameter estimation
- demographic stochasticity.

Most importantly, there is a vast tradition of applying this matrix framework to managing endangered and threatened species (e.g., Crouse et al., 1987; Crowder et al., 1994; Doak et al., 1994; Horvitz and Schemske, 1995) with a rich underlying statistical and mathematical theory on which to draw (Caswell, 1989). Given the pace with which NMFS must make progress, it is a tremendous advantage to adopt such a standard tool without having to invent any new analytical machinery.

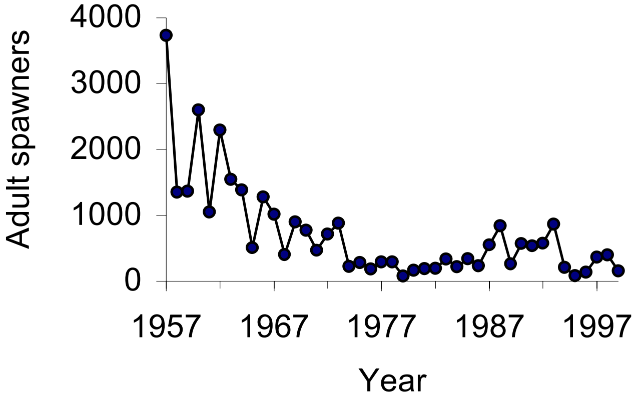
8.3.1 Estimating Matrices for Snake River Spring/Summer Index Stocks

Seven index stocks of Snake River spring/summer chinook salmon have been intensively monitored since the late-1950s (Beamesderfer et al., 1998, Table 8-3). All stocks are declining (Figure 8-7), and current spawning populations average less than 10 percent of their 1950 levels (Beamesderfer et al., 1998). In fact, these stocks appear to have an increasing rate of decline in recent years (see Section 8.2.1.3); therefore, data from the 1990 to 1994 brood years were used to parameterize simple, demographic projection matrices for these stocks (Tables 8.4, Kareiva et al., 2000). Given the apparent decline in productivity, using only these later years is a precautionary approach to evaluating threatened and endangered species. These simple projection matrices are density-independent. Again, there is little evidence supporting a density-dependence in these stocks (see Section 8.2.1.3). In fact, simple regressions of ln(recruits-per-spawner) versus spawners (as in a Ricker function) describe less of the variation than regressions of ln(recruits-per-spawner) versus time (Figures 8-7 and 8-8).

Table 8-3. Number of Spawners (S) (minus jacks) Estimated From Redd Counts and the Number of Recruits (R) to the Spawning Grounds for the Six Stocks From 1957 to 1990

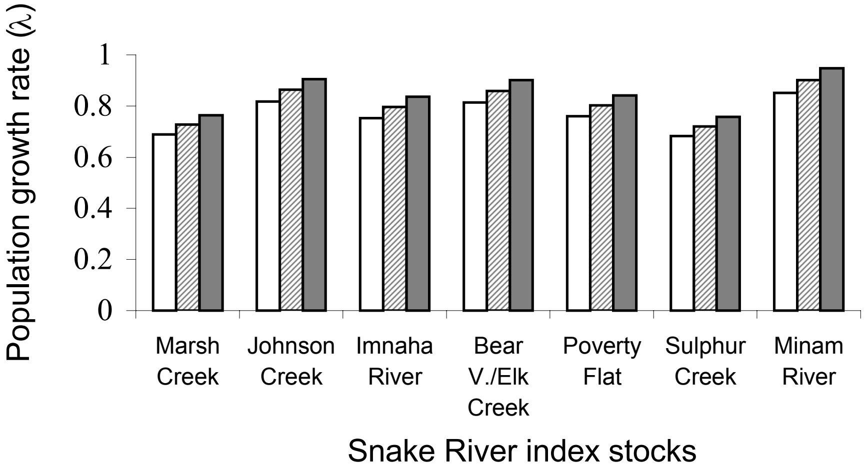
	Marsh		Johnson		Imnaha		Bear Valley		Poverty Flats		Sulphur	
yr	**S**	**R**	**S**	**R**	**S**	**R**	**S**	**R**	**S**	**R**	**S**	**R**
57	809	695	700	390	3,462	865	1,836	1,872	3,735	1,556	626	467
58	463	756	245	759	1,212	1,183	1,163	2,280	1,351	2,940	215	674
59	155	1,142	591	662	553	753	1,455	1,564	1,366	1,847	190	290
60	506	834	1,114	670	1,564	1,331	1,165	1,285	2,601	1,791	182	159
61	933	666	313	276	944	1,014	2,138	1,676	1,052	1,131	563	612
62	604	1,255	562	635	1,171	774	1,574	1,885	2,291	1,756	409	869
63	651	675	466	436	544	1,535	1,936	1,337	1,546	1,040	611	594
64	1,259	691	664	422	1,183	1,067	1,716	1,569	1,385	893	179	887
65	686	783	134	619	898	1,376	838	1,527	511	1,473	101	780
66	724	561	202	380	968	966	1,851	618	1,279	775	845	471
67	1,099	558	637	565	1,038	2,344	1,439	682	1,017	747	724	451
68	830	1,013	235	789	1,185	2,521	1,820	1,625	401	1,075	725	587
69	390	329	593	325	1,441	1,374	1,198	384	904	477	731	200
70	829	467	253	309	875	1,205	1,122	781	774	356	508	347
71	490	87	411	166	1,637	436	476	257	469	276	331	88
72	555	80	533	74	1,649	552	760	155	717	143	425	28
73	934	609	652	434	2,584	2,446	1,371	1,001	884	645	477	418
74	382	92	261	80	1,377	221	420	216	224	101	181	94
75	358	17	173	23	740	214	698	52	284	55	305	15
76	76	56	161	123	631	349	217	77	184	254	75	25
77	178	118	198	112	711	550	385	145	290	234	30	38
78	491	70	284	175	2,062	544	711	174	293	386	394	47
79	83	73	66	39	246	568	215	112	76	162	90	8
80	16	178	55	136	189	561	42	260	163	324	12	44
81	114	199	102	158	469	677	151	248	187	367	43	300
82	71	228	93	136	611	521	83	413	192	264	17	150
83	60	484	152	391	450	664	171	1,210	337	1,192	49	615
84	100	60	36	113	574	167	137	89	220	250	0	59
85	197	86	178	94	721	142	295	146	341	289	62	117
86	171	102	129	208	479	172	224	229	233	821	385	252
87	268	56	175	106	448	76	456	154	554	474	67	42
88	395	274	332	442	606	424	1,109	715	844	1,040	607	261
89	80	25	103	90	193	142	91	75	261	314	43	17
90	101	4	141	17	169	51	185	18	572	76	170	4

Note: Data were compiled by PATH.



Note: Data are based on redd (nest) counts made along a standardized segment of each stream and extrapolated to the full length (Beamesderfer et al., 1998). Poverty Flat is presented because it exhibited the median predicted rate of population growth.

Figure 8-7. Total Adult (4- and 5-year-old) Spawners from 1957 to 1999 for Poverty Flat Index Stock of Salmon River Spring/Summer Chinook Salmon



Note: Baseline matrices (clear columns) were adjusted to simulate 100 percent survival during downstream migration (hatched columns; $z = 0$ and $s_d = 1.0$) and 100 percent survival during both downstream and upstream migration (shaded columns; $z = 0$, $s_d = 1.0$, and $s_{ms} = 1.0$).

Figure 8-8. Numerical Experiments Exploring 100 Percent Survival During In-river Migration

Table 8-4. Structure of Demographic Matrices for Female Snake River Spring/Summer Chinook Salmon (from Kareiva et al., 2000)

	1	2	3	4	5
1			$(1-\mu)s_1b_3m_3/2$	$(1-\mu)s_1b_4m_4/2$	$(1-\mu)s_1b_5m_5/2$
2	S_2				
3		s_3			
4			$(1-b_3)s_4$		
5				$(1-b_4)s_5$	

Note: s_x is the probability of survival from age (x - 1) to age x, b_x is age-specific propensity to breed, s is mortality during upstream migration, and m_x is the number of eggs/female spawner of age x.

The parameters s_2 and s were further defined as follows: $s_2 = (zs_z + (1-z)s_d)s_e$, where z is the proportion of fish transported, s_d is survival during inriver migration, s_z is survival during transport, and s_e is survival in the estuary and during entry into the ocean.

$s = 1-((1-h_{ms})s_{ms}(1-h_{sb})s_{sb})$, where h_{ms} is harvest rate in the main stem of the Columbia River, s_{ms} is survival of unharvested spawners from Bonneville Dam to their spawning basin, h_{sb} is harvest rate in the subbasin, and s_{sb} is survival of unharvested adults in the subbasin prior to spawning.

Stage-specific parameters were developed or estimated from PATH data, the published literature, and other sources (Table 8-5). Baseline matrices for all seven index stocks can be found on the web at: www.sciencemag.org/feature/data/1053311.shl.

8.3.2 Results of Matrix Analyses for Snake River Spring/Summer Chinook

The dominant eigenvalues of these matrices indicate the long-term annual rates of population change (assuming demographic rates remain constant) and all are substantially less than one.

These matrices were used as the basis to determine the effect of eliminating all migration mortality except for a small tribal harvest. (Note: While perfect survival during inriver migration is unobtainable, it is a useful numerical experiment because one goal of both dam breaching and modification of intact dams is to reduce inriver migration mortality.) It was found that if each juvenile fish that migrated downstream survived to the mouth of the Columbia and every returning unharvested adult fish survived to reach the spawning grounds, the index stocks would continue to decline (Figure 8-8). Thus, management aimed solely at improving inriver migration survival cannot reverse the Snake River spring/summer chinook salmon decline (Kareiva et al., 2000).

The effectiveness of three past management actions was also evaluated: 1) reductions of harvest rates, from approximately 50 percent in the 1960s to less than 10 percent in the 1990s (Beamesderfer et al., 1998); 2) engineering improvements increasing juvenile downstream migration survival rates from approximately 10 percent, just after the last turbines were installed, to 40 to 60 percent in most recent years (Williams et al., in press); and 3) transportation of approximately 70 percent of juvenile fish from the uppermost dams to below Bonneville Dam, the lowest dam on the Columbia River (Marmorek et al., 1998). If such improvements had not been made, rates of decline would likely have been 50 to 60 percent annually (Figure 8-9), and spring/summer chinook salmon could have disappeared from the Snake River. Hence, past management actions have reduced inriver mortality, but have not reversed population declines (Kareiva et al., 2000).

Table 8-5. Parameter Values Used in Baseline Matrix Developed for Poverty Flat Index Stock of Snake River Spring/Summer Chinook Salmon (from Kareiva et al., 2000).

Parameter		Value	Reference
S_1		0.022	*Note 1*
s_2	z	0.729	*NMFS, 2000a*
	s_z	0.98	*NMFS, 2000a*
	s_d	0.202	*NMFS, 2000a*
	s_e	0.017	*Note 2*
s_3, s_4, s_5		0.8, 0.8, 0.8	*Note 3*
b_3, b_4, b_5		0.013, 0.159, 1.0	*Note 4*
μ	h_{ms}	0.020	*Beamesderfer et al., 1998*
	s_{ms}	0.794	*NMFS, 2000a*
	h_{sb}	0	*Beamesderfer et al., 1998*
	s_{sb}	0.9	*Beamesderfer et al., 1998*
m_3, m_4, m_5		3257, 4095, 5149	*NWPPC, 1989*

Note: The corresponding population growth rate (λ) is 0.760.

1. Productivity of each stock, *P*, was estimated as $\sum_{t=1}^{n} \ln(R_t / N_t)/n$, where $R_t = \sum_{x=3}^{5} N_{x,t+x}$ is the number of recruits for a particular brood year, *t*; $N_{x,t+x}$ is the number of adults of age *x* that spawn *x* years after the brood year; and *n* is the number of data years used. s_1 was found by simultaneously solving the Euler equation

$$(1-\mu)\sum_{x=1}^{5} l_x (m_x/2) b_x \lambda^{-x} = 1$$

and $l^T = e^P$, where the generation time,

$$T = (1-\mu)\sum_{x=1}^{5} l_x (m_x/2) b_x \lambda^{-x}$$

(Ratner et al., 1997).

2. To calculate se, annual counts of smolts for the aggregate run of Snake River spring/summer chinook salmon made at Lower Granite Dam were used (C. E. Petrosky and H. Schaller in PATH Weight of Evidence Report, D. Marmorek and C. Peters [eds] [ESSA Technologies, Vancouver, British Columbia, 1998], submission 10). All estimated mortality occurring below this dam until spawning was removed, and remaining mortality attributed to the period when salmon enter the estuary and nearshore ocean.

3. No direct estimates of adult survival in the ocean exist for this ESU. We set s3 = s4 = s5 = 0.8 (Ricker, 1976).

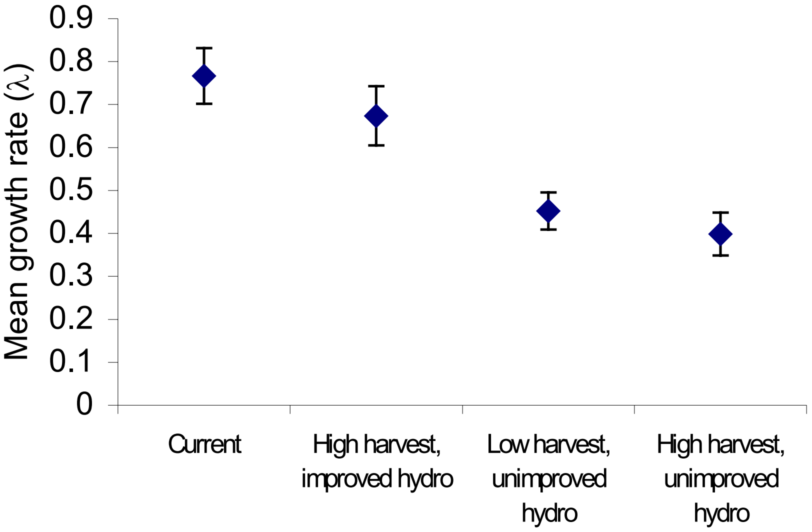
4. To find fx, the fraction of spawners of age x for females only, Kareiva et al. (2000) multiplied annual age frequencies of spawners (8) by the proportion of females at age (Hall-Griswold and Cochnauer, 1988; White and Cochnauer, 1989; Elms-Cockrom, 1998), rescaled so the frequencies summed to one, and averaged across the time series. Because these stocks rarely breed beyond age 5, Kareiva et al. (2000) set b5 = 1. Kareiva et al. (2000) estimated b3 and b4 by solving a set of simultaneous equations:

$$f_x = b_x l_x \Big/ \sum_{i=1}^{x} b_i l_i$$

for x = 1 to 5, where

$$l_x = \prod_{i=1}^{x} p_i$$

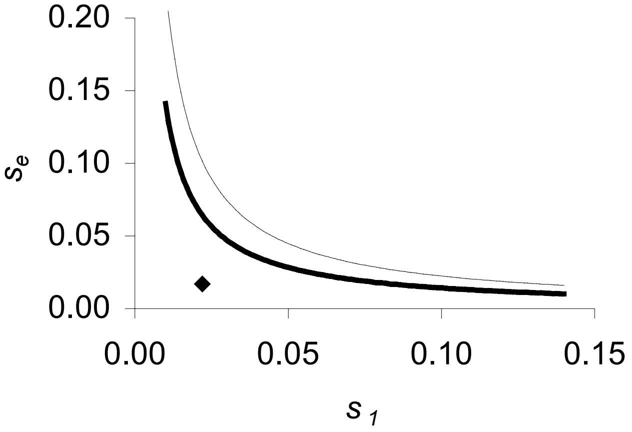
, p1 = s1, and px = (1-bx-1)sx for x > 1 (7).



Note: "Unimproved hydro" assumes current conditions, except no transportation, of juvenile fish (*z* = 0), and survival through the hydrosystem is set at rates estimated for 1977 to 1979 (s_d = 0.095 and average s_{ms} = 0.511; 5, 8); "High harvest" assumes current conditions, except harvest rates from 1960 to 1970 are used (average h_{ms} = 0.390 and average h_{sb} = 0.115; 9). Error bars are ± 1 SD.

Figure 8-9. Effectiveness of Past Management Actions Targeting In-river Survival of Snake River Spring/Summer Chinook Salmon

Finally, it is possible that improved survival in other life stages could reverse the population declines. Choosing the matrix with the median dominant eigenvalue (Poverty Flat) as a benchmark, combinations of first year survival (s_1) and early ocean/estuarine survival (s_e) values were calculated that give a dominant eigenvalue of 1.0 (a steady-state population in a deterministic world; Figure 8-10). For Poverty Flat, management actions that would reduce mortality during the first year by 6 percent, or reduce early ocean/estuarine mortality by 5 percent, would be sufficient. If reductions in mortality are simultaneously accomplished in both the first year of life and the early ocean/estuarine stage, then the combinations of mortality reductions required to produce an eigenvalue ≥ 1.0 are as modest as a 3 percent reduction in first-year mortality and a 1 percent reduction in estuarine mortality. Data to parameterize a stochastic matrix model are lacking; however, deterministic models consistently overestimate the long-run growth rates experienced in a variable environment (Caswell, 1989). Thus, a deterministic growth rate considerably greater than 1.0 is desirable. To achieve a 10 percent annual growth rate ($\lambda = 1.1$), first-year mortality must be reduced by 11 percent or early ocean/estuarine mortality must be reduced by 9 percent (Kareiva et al., 2000). Adult mortality in the ocean is neglected because ocean harvest is negligible on these stocks and management opportunities for enhancing open ocean survival are limited (Marmorek et al., 1998).



Note: Target λ = 1.0 (thick line) and 1.1 (thin line). To produce isoclines, s_1 was incrementally increased and values of s_e were searched for the smallest value causing l to exceed the target λ. Current parameter values are shown for reference.

Figure 8-10. Isoclines Calibrating Improvements in s_1 and s_e for Poverty Flat Index Stock of Snake River Spring/Summer Chinook Salmon

The challenge of increasing first year and estuarine survival shifts scientific inquiry from demographic modeling to identifying management actions that may produce the desired improvements. Because Snake River spring/summer chinook salmon spawn in the upper reaches of Snake River tributaries, dam breaching is unlikely to affect available spawning habitat or first year survival, but could improve estuarine survival considerably. Although survival of juvenile fish during barging is high, barging may reduce the subsequent survival of barged fish relative to those that swim downstream. Breaching the lower Snake River dams would mean the end of fish transportation operations and would, therefore, eliminate any delayed mortality from transportation. Additionally, the removal of four of the eight dams encountered by Snake River salmon may increase the physiological vigor of salmon that swim down river, thus improving survival during the critical estuarine phase. If this indirect mortality was 9 percent or higher, then dam breaching could reverse the declining trend of Snake River spring/summer chinook salmon (Figure 8-11). Unfortunately, estimating the magnitude of any indirect mortality from passage through the lower Snake River dams is difficult because identifying fish as appropriate “controls” for the potential effects of these dams is problematic. Additionally, if the lower Snake River dams were removed, the fish would still have to negotiate four Columbia River dams, and baseline mortality would still include any indirect mortality attributable to passage through those dams.

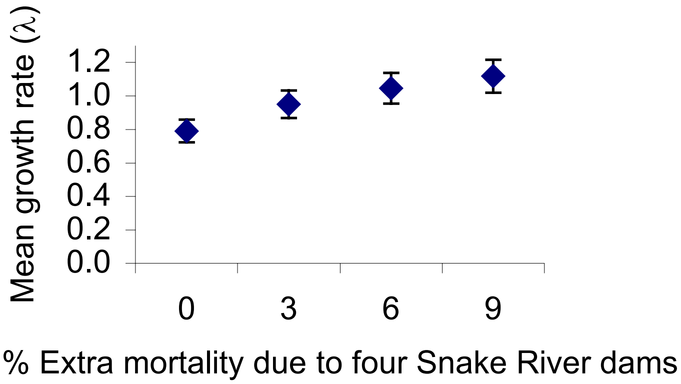


Figure 8-11. Potential Effects of Dam Breaching for Snake River Spring/Summer Chinook Salmon Index Stocks

In addition to straightforward improvements in migration survival, breaching the four lower Snake River dams might also improve survival in postmigration stages. Delayed transportation mortality is conventionally measured as D, a ratio of survival of transported fish relative to nontransported fish; the current best estimate for this ESU is $D = 0.7$ (transported fish survive at 70 percent the rate of nontransported fish) (NMFS, 2000a). Extra mortality results from the physiological stress of passing through dams. Baseline mortality (m) is increased by a percentage, e, such that mortality observed in the estuary today is $m + e*m$. If the four lower Snake River dams were breached, the hypothesized e would go to zero, causing s_e to increase. For this figure no fish transportation was assumed ($z = 0$), improved survival during downstream ($s_d = 0.607$) and upstream migration ($s_{ms} = 0.913$), $D = 0.7$, and that the "extra mortality" indicated along the ordinate axis becomes zero, corresponding to the following values: $s_e = 0.022$ for $e = 0$ %, $s_e = 0.052$ for $e = 3$ %, $s_e = 0.082$ for $e = 6$ %, and $s_e = 0.112$ for $e = 9$ %.

8.3.3 Additional Details about Matrix Analyses

Matrices reflecting so-called average conditions can be calculated in many different ways. The matrices in Kareiva et al. (2000) used median recruits per spawner rates (see above). Alternatively, mean recruits per spawner, or the geometric mean matrix, could be used. All three approaches were tried, and the results discussed below are not qualitatively altered by these alternative methods for taking an average. For a detailed population viability analysis, separate estimates of temporal variation for each matrix entry, as well as some estimate of how the different matrix entries covary, would be warranted. There is little chance that such detailed data will be forthcoming for ANY salmonid stock over the next 10 years. Arguably, it is also unlikely that much would be gained from these more detailed data, except slightly more refined estimates of extinction risks. This is not where NMFS believes future research needs to be directed.

8.4 Estimating Projection Matrices for Fall Chinook Salmon and Management Experiments

Snake River fall chinook differ from Snake River spring/summer chinook in three important ways: 1) fall chinook are ocean-type salmonids, migrating to the ocean during their first year of life; 2) fall chinook return to spawn at ages 2 (jacks), 3, 4, 5, and 6, whereas the 7 spring/summer index stocks return only at ages 3, 4, and 5; and 3) fall chinook are subjected to considerable ocean harvest, whereas there is virtually no ocean harvest of the spring/summer stocks. The demographic matrix for fall chinook is therefore a six-by-six matrix, with ocean harvest factored into the adult survival terms (see below).

To derive parameter estimates for Snake River fall chinook, NMFS used annual counts of natural-origin jacks and adults at the uppermost dam (1980 to present) and age frequencies of spawners based on year-specific proportion at age calculated from Lyons Ferry Hatchery fall chinook CWTs (Peters et al., 1999). Mainstem harvest, ocean harvest, and Bon to Basin conversion rates were also obtained from Peters et al. (1999). For harvest rates and survival during upstream migration, data from 1993 to 1996 were used because there were reductions in harvest starting in 1993 under ESA management. Although there are potential problems involved with using data from hatchery fish, the best available information on age-specific fecundity and sex ratio at age come from fish at Lyons Ferry Hatchery (Mendel et al., 1996).

Age-specific parameters used in Snake River fall chinook analyses are shown in Table 8-6.

Table 8-6. Age-Specific Parameters Used in Snake River Fall Chinook Analyses

	2	**3**	**4**	**5**	**6**
Age frequency of females (f_x)	0	0.129	0.652	0.198	0.020
93-96 Ocean harvest rate (h_x)	0.0123	0.0465	0.1368	0.1838	0.1953
Female eggs per female spawner (m_x)		1442.5	1566.5	1625.5	1625.5
Propensity to breed (b_x) (solved as in Appendix A)	0	0.081	0.648	0.859	1.0
1993 through 1996 Mainstem adult harvest rate		0.174			
1993 through 1996 adult Bon to Basin conversion rate		0.471			
s_1 (solved as in Appendix A worksheet)		0.0102			

These parameters are then substituted into the following matrix where, as previously, μ= 1-(0.9 * Bon to Basin * (1-mainstem harvest).

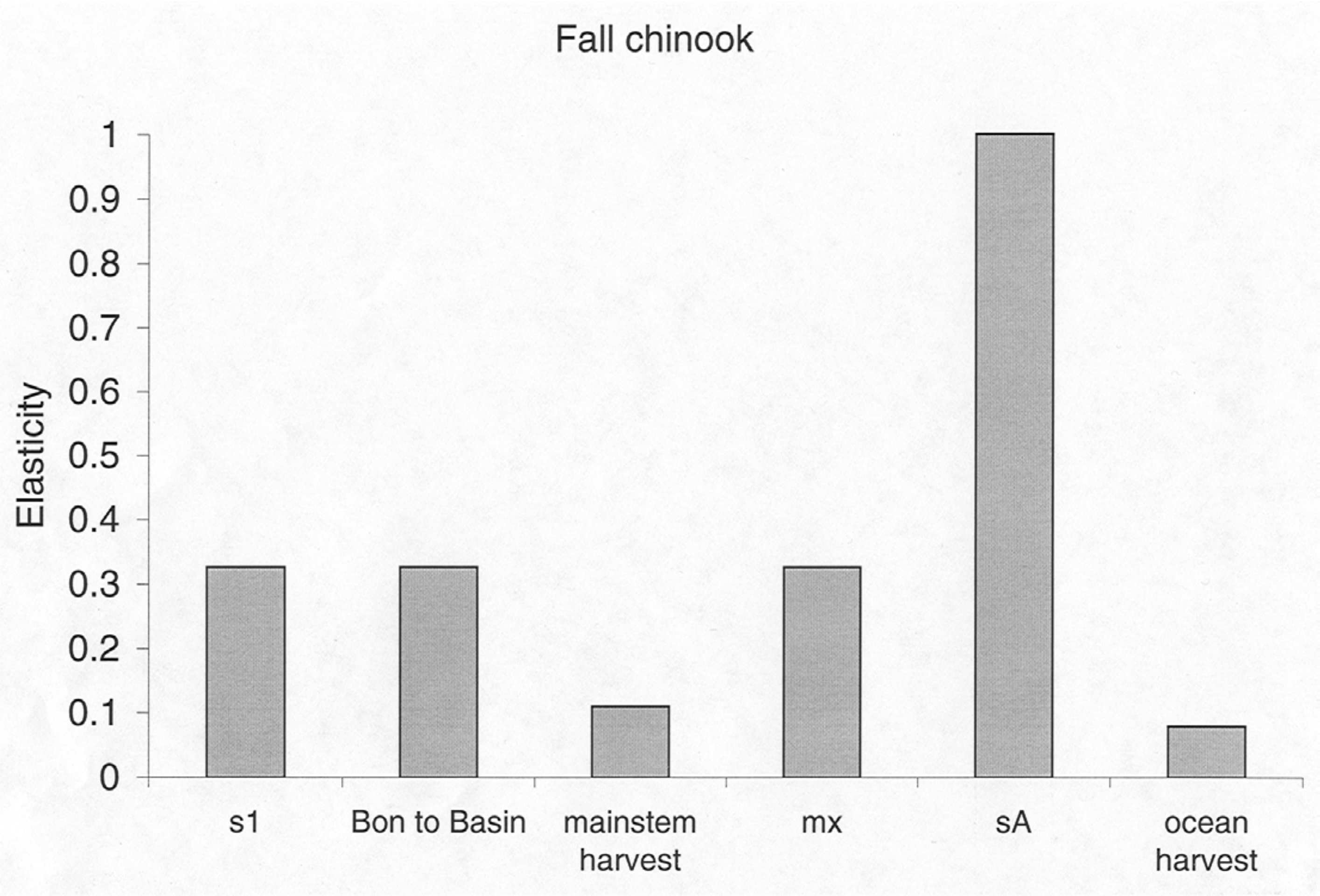
$$\begin{array}{ccccccc}
 & 1 & 2 & 3 & 4 & 5 & 6 \\
1 & 0 & 0 & (1-\mu)s_1b_3m_3 & (1-\mu)s_1b_4m_4 & (1-\mu)s_1m_5 & (1-\mu)s_1m_6 \\
2 & (1-h_2)s_A & 0 & 0 & 0 & 0 & 0 \\
3 & 0 & (1-h_3)\ s_A & 0 & 0 & 0 & 0 \\
4 & 0 & 0 & (1-b_3)(1-h_4)s_A & 0 & 0 & 0 \\
5 & 0 & 0 & 0 & (1-b_4)(1-h_5)s_A & 0 & 0 \\
6 & 0 & 0 & 0 & 0 & (1-b_5)(1-h_6)s_A & 0
\end{array}$$

Data regarding survival during downstream migration and the proportion of smolts transported are generally much poorer for fall chinook than for spring/summer chinook. Therefore, s_1 includes everything from egg hatch, downstream migration, and survival in the estuary and entry into the ocean environment. Due to the lack of data, no attempt was made to break s_1 down into all of its component pieces.

The sensitivity of the matrix for fall chinook was evaluated in two ways: 1) elasticity analysis and 2) numerical experiments investigating the percentage improvement associated with saving 1 out of 10 salmon that currently die at each stage. The elasticity results for fall chinook (Figure 8-12) closely mirror those for spring/summer chinook salmon (not shown). In particular, the most sensitive parameter is the survival of adults in the ocean, again because individuals at this stage have survived periods of high mortality and are close to the age of reproduction. Results of the saving 1-of-10 experiments for fall chinook (Figure 8-13) are also similar to those for spring/summer chinook. Specifically, reducing mortality during the first year of life produces the largest change in population growth rate (Figure 8-14; recall that for fall chinook, s_1 includes survival in the estuary and entry into the ocean environment). This result can be largely attributed to the low estimated survival during the s_1 stage. Simply stated, because survival of s_1 fish is so low, saving 1 out of 10 fish that would die at this stage involves saving a great many more fish than it would for any of the other stages.

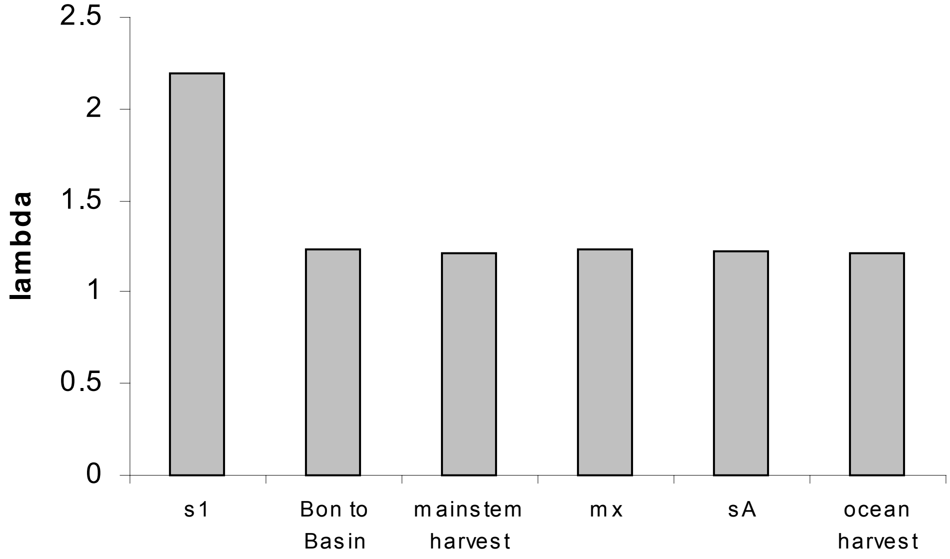
After 1993, ESA management has led to decreases in rates of harvest for Snake River fall chinook salmon. However, one potential management option would be to enforce further reductions in either ocean or mainstem harvest or both (i.e., more than those examined in the save 1-of-10 experiments). An approximately 4 percent increase in λ would be required to lower the probability of quasi-extinction within 100 years for fall chinook to less than 1 in 100. This magnitude of change could be accomplished with a 75 percent reduction in ocean harvest, a 75 percent reduction in mainstem harvest, or a 50 percent reduction in both ocean and mainstem harvest (Figure 8-15); thus, harvest reductions can yield a biologically reasonable management option for Snake River fall chinook.

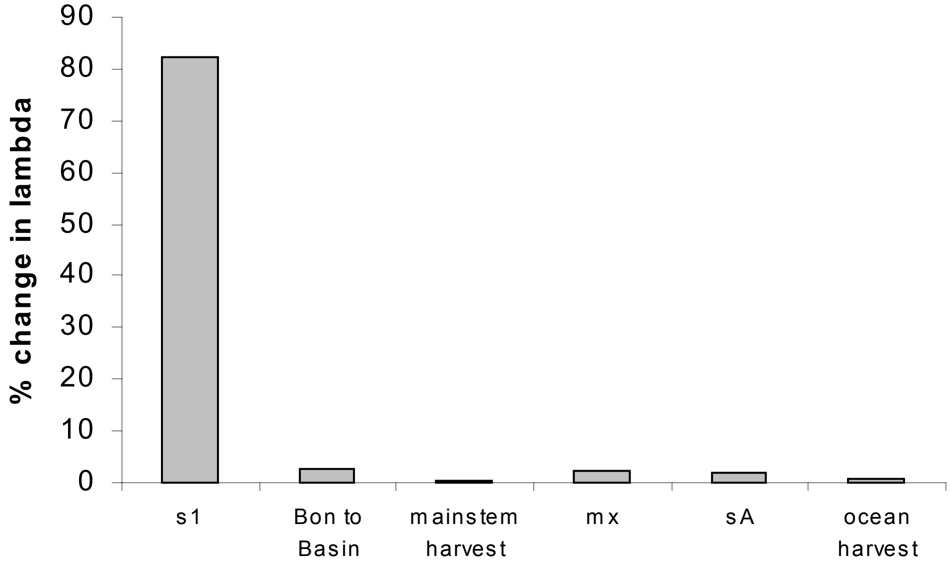
It is more difficult to assess the potential benefits of dam breaching for Snake River fall chinook salmon because data regarding survival during downstream migration and the proportion of smolts



Note: Elasticity (sensitivity of population growth rate to changes in demographic parameters) for fall chinook. s1 = survivorship to the fish's first birthday (note that this includes freshwater rearing and estuarine survival); Bon to Basin = survivorship of upstream migrants from Bonneville Dam to the Snake River Basin; mainstem harvest = mainstem harvest rate; mx = fecundity of females of age x; sA = survivorship of adults in the ocean; ocean harvest = ocean harvest rate. Survivorship of adults in the ocean has the highest elasticity because these individuals have survived periods of high mortality and are near the age of reproduction.

Figure 8-12. Sensitivity of Annual Population Growth to Small Changes in Components of Fall Chinook Salmon Demographic Projection Matrix





Note: Percent change in population growth rate with a 10 percent reduction in mortality at each life stage for fall chinook. (A 10 percent increase in fecundity was also analyzed.) s1 = survivorship to the fish's first birthday (note that this includes freshwater rearing and estuarine survival); Bon to Basin = survivorship of upstream migrants from Bonneville Dam to the Snake River Basin; mainstem harvest = mainstem harvest rate; mx = fecundity of females of age x; sA = survivorship of adults in the ocean; ocean harvest = ocean harvest rate. Fall chinook population growth rate shows the greatest sensitivity, by this measure, to reduced mortality during the first year of life (which includes freshwater rearing, and estuarine and early ocean survivorship) because these are periods during which there is very high mortality.

Figure 8-13. Improvements in Fall Chinook Salmon Annual Population Growth with 10 Percent Reductions in Mortality During Different Lifestages

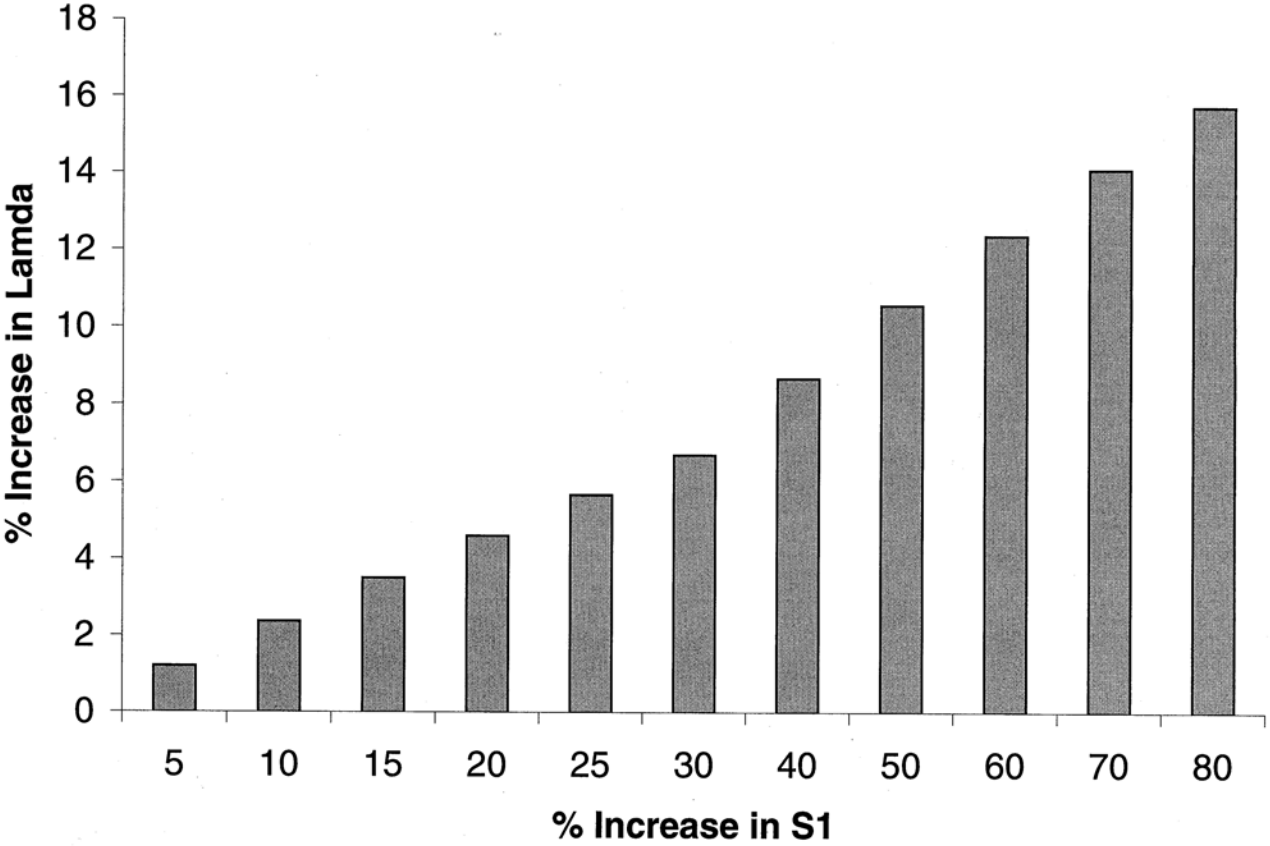


Figure 8-14. Increase in Fall Chinook Annual Population Growth with a Range of Increases in First Year Survivorship

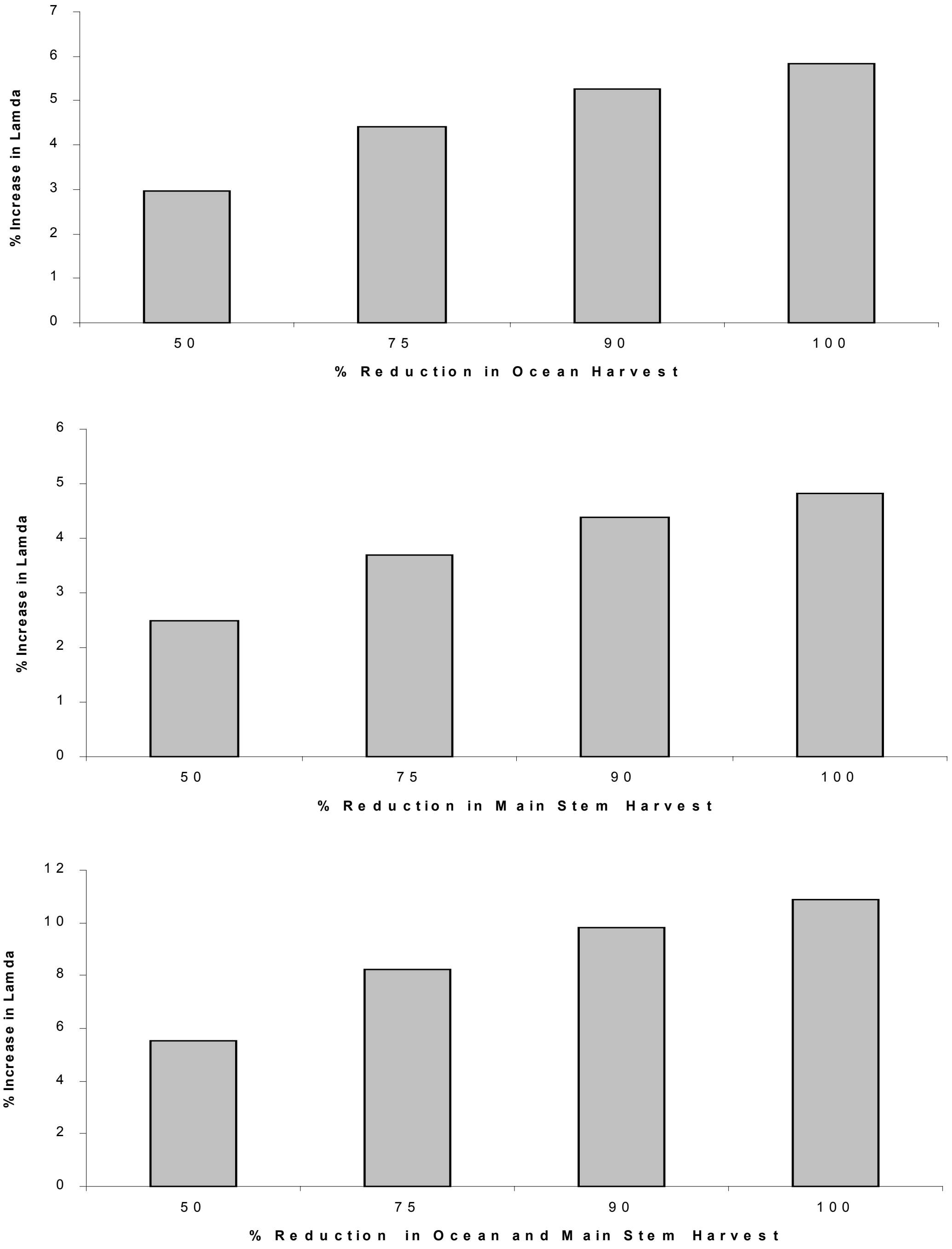


Figure 8-15. Increases in Fall Chinook Annual Population Growth with a Range of Harvest Reductions

transported are not as abundant. However, the majority of effects would likely occur in the s_1 stage, which includes both downstream migration and post-Bonneville survival in the estuarine environment (where latent effects of dams are likely to accrue). The percent increase in λ was expected to result from a broad range of potential changes in s_1 survival. Again, an approximate 4 percent increase in λ is expected to lower the probability of quasi-extinction within 100 years to 1 in 100. This level of improvement in λ could be achieved with a less than 20 percent increase in s_1. Whether or not such a change in s_1 would actually occur under dam breaching is unknown. Lastly, as noted in the PATH analysis, dam breaching would open up habitat for fall chinook salmon. Expansion of populations to fill this habitat will still require an increase in annual population growth rates above current levels.

8.5 Limitations of the CRI Analytical Framework

There are several limitations of the CRI analytical framework, just as there are limitations of the PATH analytical framework. First, CRI has not yet developed effective approaches for estimating carrying capacity; hence, while CRI analyses may be apt for populations at low density, as stocks rebuild, the analyses will need to be modified. Second, CRI cannot address questions about refinements in the hydropower systems because the hydropower system does not appear explicitly in CRI models; this means that instead of mechanistic relationships between flow regimes and survival, CRI treats flow variability as unexplained environmental variability. Third, CRI has not yet developed adequate analyses of the feasibility of achieving particular demographic improvements as a result of specific management actions. This will be the hardest challenge for CRI and represents the task that PATH has foundered on. The hope is that by isolating these feasibility studies from population projection models, the types of studies and data needed will become more apparent. It remains to be seen whether this hope is warranted. Fourth, by focusing so much on current conditions, CRI fails to incorporate potential influences of decadal oscillations in ocean conditions and infrequent catastrophes. Finally, like PATH, CRI has thus far essentially treated each population as independent and has built up its risk analyses without attention to ESU-wide meta-population structure. Many of these limitations are not necessary attributes of CRI, but rather represent its early stages of development. The challenge will be in keeping it simple and transparent, while addressing the above limitations.

8.6 Synthesis of Results Across All Salmonids

The CRI analyses attempt to put dam breaching in the context of a menu of other management actions and to account for extinction risks. From the perspective of population growth rate alone, it appears that harvest reductions (or moratoriums) would be adequate to sufficiently increase annual rates of population growth for fall chinook; it also appears that modest survival improvements due to dam breaching could accomplish the same goals. Of course, as discussed in Section 5, dam breaching would also increase the availability of habitat for fall chinook salmon and hence the carrying capacity, whereas harvest reductions have no such possibility.

The situation for spring/summer chinook is much more complicated. First of all, there is no silver bullet that is likely to adequately reduce their extinction risks. For dam breaching alone to recover spring/summer chinook salmon, very optimistic scenarios would need to be assumed about how much survival below Bonneville Dam could be improved due to the elimination of latent mortality not measured during inriver downstream and upstream migration. For aggressive habitat management and other management actions alone to be sufficient, magnitudes of habitat improvements that are not known to be achievable would have to be assumed, as well as reductions in predation impacts for which data are still scant.

9. Updates on Differential Delayed Transportation Mortality and Research Related to Dam Passage

Since the production of the draft Anadromous Fish Appendix, NMFS has produced several "white papers" on the topics of flow-survival relationships, passage survival, transportation research, and predation. These white papers, with comments and responses to those comments, can be viewed at: http://www.nwfsc.noaa.gov/pubs/nwfscpub.html.

The following description of the most recent estimates of *D*, or differential delayed transportation mortality, is an excerpt from the Transportation White Paper.

THE BENEFITS OF TRANSPORTATION: THE *D* CONCEPT

The current configuration of juvenile bypass systems at dams on the lower Snake and Columbia rivers provides the option of transporting spring/summer chinook salmon and steelhead from three locations on the lower Snake River (Lower Granite, Little Goose, and Lower Monumental Dams) and from McNary Dam on the Columbia River. In most recent years, the general (nonPIT-tagged) downstream migrant population collected during the spring was transported from lower Snake River dams, but transportation was discontinued from McNary Dam after the 1994 outmigration. For fish collected at a dam, transportation is generally the preferred option when the expected adult return rate of fish transported to below Bonneville Dam exceeds the expected return rate of fish that remained in the river to migrate downstream through the hydropower system.

For a given dam, the smolt-to-adult return rates (SARs) for transported and inriver fish are each composed of two components: the survival from the collection dam to below Bonneville Dam, and the survival from below Bonneville Dam to adult return, referred to as "post-Bonneville Dam" survival. The SARs can be described by the equations

$$SAR_T = S_{d,T} \cdot S_{pb,T}$$

and

$$SAR_I = S_{d,I} \cdot S_{pb,I}$$

where the subscripts *T* and *I* refer to transported and inriver fish, respectively; S_d is downstream survival, and S_{pb} is the post-Bonneville survival. One reason to split the SARs into two components is that S_d (downstream survival) can be estimated, whereas S_{pb} currently cannot be estimated directly but must be inferred from SARs and downstream survival estimates.

By comparing post-Bonneville survival of transported fish to inriver fish, the question can be addressed of whether transported fish survive as well after they are released as do their inriver counterparts. “Differential post-Bonneville Dam survival” has been termed *D* and is expressed by the following equation:

$$D = \frac{S_{pb,T}}{S_{pb,I}}$$

If transported fish and inriver fish have the same survival from the transport release site to return as adults, then $D = 1.0$. If transported fish incur greater mortality after release from the barge, then $D < 1.0$.

Based on the equations above, the familiar *T:I* ratio (ratio of the SARs) can be expressed as

$$T{:}I = \frac{SAR_T}{SAR_I} = \frac{S_{d,T}}{S_{d,I}} \cdot \frac{S_{pb,T}}{S_{pb,I}} = \frac{S_{d,T}}{S_{d,I}} \cdot D$$

Transportation benefits fish stocks from a particular location only if the SAR for transported fish exceeds that for inriver fish; that is, if the *T:I* ratio exceeds 1.0. Because $S_{d,T}$ (survival in the barge from the collection dam to below Bonneville Dam) is near 1.0, the decision essentially reduces to a comparison of survival to below Bonneville for fish that migrate in the river versus differential post-Bonneville Dam survival. In terms of the equations, transportation benefits fish only if $D > S_{d,I}$.

One consequence of this relationship is that if *D* is the same for each transportation site, then the benefit of transportation is greater for collection sites farther upstream. This is because $S_{d,I}$ increases for sites farther downstream. This follows from the common-sense deduction that fish transported from Lower Granite Dam avoid more direct inriver mortality than fish transported from McNary Dam.

Estimates of *D* for Snake River ESUs

Below, estimates are presented of *D* for Snake River spring/summer chinook salmon and steelhead derived from PIT-tag data. A discussion of *D* estimation for Snake River subyearling fall chinook salmon is also included.

For spring/summer chinook salmon and steelhead, annual estimates of *D* were based on T:I ratios for wild fish PIT-tagged above Lower Granite Dam. The inriver control group for a given year was composed of fish that represented the unmarked population (it did not include PIT-tagged fish bypassed back to the river at dams where the general migrant population was transported). Thus, the control group was composed only of nondetected fish at lower Snake River dams and at McNary Dam in 1994, and of nondetected plus fish detected only at McNary Dam in 1995 through 1997. In the transport group, SARs for fish transported from different dams were weighted proportionally to the estimated proportion of nontagged fish transported from each dam, so that transported PIT-tagged fish were representative of the transported nontagged population at large. Estimates of *D* also depended on estimates of reach-specific survival between Lower Granite Dam and Bonneville Dam (Muir et al., in review; Sandford and Smith, in review; and Williams et al., *submitted*), survival from barge-loading to below Bonneville Dam for transported fish (assumed 0.98 for all dams in all years), and estimates of detection probabilities at collector dams. Detections of PIT-tagged fish were used to estimate survival between the tailraces of Lower Granite and McNary Dams in all years. Estimates of survival between the tailraces of McNary and Bonneville Dams were extrapolated from estimates of survival between Lower Granite Dam and McNary Dam for years when direct survival estimates were not available. Two extrapolation methods were used: 1) per-project survival between McNary and Bonneville Dams (three projects) was assumed equal to per-project survival between Lower Granite to McNary Dams (four projects), and 2) per-kilometer survival between McNary and Bonneville Dams (236 kilometers) was assumed equal to per-kilometer survival between Lower Granite to McNary Dams (225 kilometers). Empirical survival estimates between McNary and Bonneville Dams were possible for steelhead from 1997 to 1999 and for spring/summer chinook salmon in 1999. Comparison of extrapolation methods to empirical estimates was inconclusive: per-kilometer extrapolation was closer to the empirical estimate in three of four cases, and per-project extrapolation was closer once.

For PIT-tagged wild fish of the two species, Tables 9-1 and 9-3 (based on per-project extrapolations for the lower river) and Tables 9-2 and 9-4 (based on per-kilometer extrapolations for the lower river) provide estimated SARs for transport and control groups, inriver survival estimates, and estimates of *D* with confidence intervals for each year. In addition, the geometric mean of the annual point estimates of *D* was calculated across years. All estimated SARs represent the proportion of smolts that left Lower Granite Dam and returned to Lower Granite Dam as adults.

The estimates of *D* are derived from estimated numbers of smolts in various passage history categories from analyses by Sandford and Smith (in review). Based on peer review of the first submitted draft of that manuscript, refinement of estimation methods is currently taking place. Slightly different estimates of *D,* based on previous versions of Sandford and Smith's document have been distributed elsewhere (e.g., Draft Anadromous Fish Appendix, Corps, 1999). Methods for combining passage history categories to represent the population at large have also been refined since the first estimates of *D* were calculated. Furthermore, all the methods are subject to further revision, though only small effects on *D* estimates are expected. In general, estimates of *D* have varied little relative to the precision (width of confidence intervals) of the estimates. For example, in all iterations, the geometric mean of 1994 to 1996 estimates for wild spring/summer chinook salmon was between 0.78 (Table 9-1) and 0.83.

Table 9-1. Estimates of *D* (per Project Expansion) for Wild Snake River Spring/Summer Chinook Salmon (1994 through 1997)

Year	SAR_T (adults)	SAR_I (adults)	Surv.	*D* (95% C.I.)
1994	0.52 (13)	0.25 (6)	0.335	0.85 (0.01, 1.69)
1995	0.30 (8)	0.33 (10)	0.557	0.55 (0.03, 1.06)
1996	0.52 (2)	0.24 (5)	0.469	1.02 [(0 .69), 2.72]
1997	2.46 (4)	2.05 (17)	0.474	0.61 [(0.08), 1.29]
geometric mean 1994 through 1997:				**0.73**

Notes: SAR_T is the estimated SAR for transported fish. SAR_I is the estimated SAR for inriver (control) fish. Total adult returns ( ) are provided for all estimated SARs. Surv. is the estimated survival from Lower Granite Dam to Bonneville Dam for inriver fish (per-project extrapolation). *D* is estimated for each year (along with approximate 95 percent confidence intervals), and the geometric mean of the yearly *D* is provided. (1997 returns incomplete.)

Table 9-2. Estimates of *D* (per Project Expansion) for Wild Snake River Steelhead (1994 through 1997)

Year	SAR_T (adults)	SAR_I (adults)	Surv.	*D* (95% C.I.)
1994	1.29 (8)	1.16 (6)	0.416	0.51 [(0.04), 1.06]
1995	0.40 (1)	0.00 (0)	0.583	NA
1996	0.59(1)	0.58 (4)	0.531	0.54 [(0.68), 1.76]
1997	0.82 (3)	0.57 (3)	0.474	0.71 [(0.45), 1.87]
geometric mean 1994, 1995, 1997:				**0.58**

Notes: SAR_T is the estimated SAR for transported fish. SAR_I is the estimated SAR for inriver (control) fish. Total adult returns ( ) are provided for all SARs. Surv. is the estimated survival from Lower Granite Dam to Bonneville Dam for inriver fish (per-project extrapolation). *D* is estimated for each year (along with approximate 95 percent confidence intervals), and the geometric mean of the yearly *D* is provided.

Table 9-3. Estimates of *D* (per Kilometer Expansion) for Wild Snake River Spring/Summer Chinook Salmon (1994 through 1997)

Year	SAR_T (adults)	SAR_I (adults)	Surv.	*D* (95% C.I.)
1994	0.52 (13)	0.25 (6)	0.260	0.66 (0.01, 1.31)
1995	0.30 (8)	0.33 (10)	0.501	0.49 (0.02, 0.96)
1996	0.52 (2)	0.24 (5)	0.412	0.89 [(0 .60), 2.39]
1997	2.46 (4)	2.05 (17)	0.417	0.54 [(0.07), 1.14]
			geometric mean 1994 through 1997:	**0.63**

Notes: SAR_T is the estimated SAR for transported fish. SAR_I is the estimated SAR for inriver (control) fish. Total adult returns ( ) are provided for all SARs. Surv. is the estimated survival from Lower Granite Dam to Bonneville Dam for inriver fish (per-kilometer expansion). *D* is estimated for each year (along with approximate 95 percent confidence intervals), and the geometric mean of the yearly *D* is provided. (1997 returns incomplete)

Table 9-4. Estimates of *D* (per Kilometer Expansion) for Wild Snake River Steelhead (1994 through 1997)

Year	SAR_T (adults)	SAR_I (adults)	Surv.	*D* (95% C.I.)
1994	1.29 (8)	1.16 (6)	0.336	0.41 [(0.04), 0.86]
1995	0.40 (1)	0.00 (0)	0.528	NA
1996	0.59(1)	0.58 (4)	0.476	0.49 [(0.61), 1.58]
1997	0.82 (3)	0.57 (3)	0.474	0.71 [(0.45), 1.87]
			geometric mean 1994, 1995, 1997:	**0.52**

Notes: SAR_T is the estimated SAR for transported fish. SAR_I is the estimated SAR for inriver (control) fish. Total adult returns ( ) are provided for all SARs. Surv. is the estimated survival from Lower Granite Dam to Bonneville Dam for inriver fish (per-kilometer expansion). *D* is estimated for each year (along with approximate 95 percent confidence intervals), and the geometric mean of the yearly *D* is provided.

Adult returns of wild Snake River salmonids PIT-tagged above Lower Granite Dam were particularly small, yielding large confidence intervals about the yearly estimates. Thus, the above *D* estimates should be viewed with caution. Much more data will be necessary before more reliable and more meaningful *D* estimates can be calculated.

It is not surprising that survival of transported fish in the post-Bonneville phase is generally not as high as that of inriver fish. First, passage through reservoirs and dams likely culls weaker downstream migrants, with only the stronger fish surviving to below Bonneville Dam. Transported fish face no physical obstacles and are generally released below Bonneville Dam within 36 to 48 hours after collection. The culling process for them likely continues after release. Moreover, some fish arriving at the hydropower system are certain to die (i.e., fish with active or advanced bacterial kidney disease infections) during the ensuing 3-week period whether they migrate through the hydropower system or are transported. These fish would die even if the hydropower system were not in place. Survival estimates of inriver fish account for this mortality. If transported, these fish would not die until after release below Bonneville Dam. Finally, high fish densities on barges may cause stress and promote horizontal disease transmission, either of which could result in greater mortality after release than the inriver migrants.

For Snake River fall chinook salmon, a great deal of uncertainty exists regarding the value of *D*. This is primarily because no formal transportation studies have been performed for these fish, and thus the empirical basis for *D* estimates is not as strong as for spring migrants. Estimates of *D* require multiple assumptions, which are usually model-based. In addition, transportation methods have changed through the years, from fish being released near the bank of the river in areas known to have concentrations of predators (1993 and before) to being released in the middle of the river at varying locations (1994 and after). Further, transportation modes may change in the future from primarily trucked-based to more reliance on barges (there is concern that trucked fish do not have the opportunity for imprinting and may be prone to straying).

The PATH analysis of Snake River fall chinook salmon (Peters et al., 1999) used several methods to estimate *D*, each with inherent strengths and weaknesses. The first method was to estimate *D* from spawner-recruit data by incorporating *D* as a "free" parameter in a life-cycle model. This resulted in a wide range of values with a median value of about 0.05. However, the estimate of *D* is confounded by other parameter estimates, notably *E*, the spawning effectiveness of hatchery strays. The second method involved estimating *D* based on PIT-tagged fish (primarily hatchery origin), some of which were known to have been transported. For migration year 1995, this resulted in a *D* estimate of approximately 0.24. This estimate represented only one year (although the method could be used to estimate *D* for 1996), and because sample sizes were small the estimate had a large confidence interval. A third source of information is transportation studies conducted on subyearling chinook salmon (primarily Hanford Reach fish) at McNary Dam during the years 1978 through 1983. T/Cs for these studies were relatively large, and resulting *D* estimates were generally greater than 1.0. These results were obtained primarily from a different stock than Snake River fall chinook salmon, using different transportation operations. However, they may represent higher *D* values than possibly could be achieved with improved transportation operations in the future. Hopefully, transportation studies will be initiated during the 2000 outmigration to improve our understanding of *D* for Snake River fall chinook salmon.

10. Summary of Results, Uncertainties, and Opportunities for Resolving the Uncertainties

10.1 The Bottom Line of the PATH Analyses

In the PATH analyses, dam breaching causes a larger fraction of simulated future fish populations to exceed survival and recovery criteria than any other hydropower management option. These computer projections are quantitative for spring/summer chinook salmon and fall chinook salmon, and are qualitative for steelhead. Sockeye salmon are so depleted that no analysis is possible. The critical uncertainty in this PATH conclusion is the assumption that transportation of fish in barges leads to a significant differential delayed transportation mortality after the fish are released below Bonneville Dam, or that passage through the hydropower system by nontransported fish causes a significant extra mortality after the fish have passed Bonneville Dam and moved into the estuary and ocean. In general, PATH analyses produce quite optimistic predictions for recovery if dams are breached; for example, under 100 percent of PATH assumption sets, spring/summer chinook salmon are predicted to achieve recovery within 48 years if the dams are breached (Table 2-2.4-3 of PATH 1998 Final Report, Marmorek et al. [1998]). The management scenario corresponding to maximizing transportation and other hydropower system improvements is much less likely to yield recovery according to PATH analyses, but still has some marked chance of success on its own (roughly a 1 in 2 chance). The PATH analyses do not allow an estimate of the risk associated with delaying action while learning more about extra mortality hypotheses and differential delayed transportation mortality.

10.2 The Bottom Line of the CRI Analyses

In general, the CRI analyses are less optimistic than PATH analyses because they indicate substantial risks of extinction and/or population decline for spring/summer chinook salmon, fall chinook salmon, and steelhead over the next 100 years if current conditions hold. The extinction calculations estimate the probability of true extinction (escapement falling to one fish in any one generation), and are therefore very conservative measures of extinction risk.

Unlike PATH, the CRI analyses suggest that no single management action is likely to result in sufficiently improved demography for spring/summer chinook salmon. For dam breaching alone to recover spring/summer chinook salmon, it would have to produce improvements in estuarine and early ocean survival substantially (from approximately 2 percent to approximately 10 percent). On a more optimistic note, the CRI analyses suggest that a combination of improvements spread throughout the life cycle, and attained by a mixture of different management actions, could promote adequate annual population growth for spring/summer chinook salmon. Numerical experiments that correspond to manipulations of "current demography" indicate that small improvements in estuarine and early ocean survival or in the survival of newly born fish, will yield the greatest rewards in terms of enhanced population growth. Moreover, if many improvements are added together, CRI analyses suggest that annual rates of population growth could be increased enough that stocks of spring/summer chinook salmon could rebuild. The management actions that might produce these demographic improvements include habitat restoration, reducing predation pressure in reservoirs and the estuary, potentially manipulating the time and release position of downstream migrants, improved water quality, mitigation of negative hatchery impacts, continued harvest restrictions, and, of course, dam breaching. But no single silver bullet solution is supported by the data when it comes to spring/summer chinook salmon.

10.3 Critical Uncertainties About the Feasibility of Attaining Required Demographic Improvements

The major uncertainty for the CRI analyses is the biological feasibility of using particular management actions to achieve sufficient demographic improvements. Harvest reductions, which are clearly and undeniably converted into survival improvements, are the one management action for which the feasibility of achieving a specific demographic effect is not contentious. In contrast, the demographic consequences of virtually every other management action are uncertain.

CRI sensitivity analyses of stage-structured demography for fall and spring/summer chinook salmon indicate that improvements in survival of fish during the first year of life before migrating downstream or during entry into the estuary and ocean are likely to have the greatest impacts on annual population growth rates. This sensitivity analysis thus points toward the need for feasibility studies aimed at how to attain improvements in survival during these key life stages. Critical uncertainties regarding the connection between management actions and improvement in fish demography or fitness are discussed below, along with specific suggestions for research that could help resolve these uncertainties.

10.3.1 Could Habitat Restoration Help Recover Threatened Snake River Salmonids?

Improved habitat conditions might lead to substantial improvements in the survival of fish during their first year of life, but a better understanding of the relationship between habitat quality and salmonid population dynamics is required. This knowledge would 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 the abundance, survival, and productivity of salmonids in the Snake River Basin? 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. Analyses by Bilby et al. (1999, Annex G) reveal that only a few subwatersheds account for the bulk of salmon productivity in any given river basin. Using this fact, it may be possible to identify the habitat features that promote productivity, as well as target particular subwatersheds that are prime candidates for restoration.

2) 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 percent of historical levels. The extent to which the elimination of this annual nutrient subsidy has contributed to the decline in salmon and steelhead populations is not known. Likewise, the extent to which these low input 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 these materials in the Snake River system may assist in developing approaches to habitat and harvest management that will contribute to recovery of these depressed stocks.

Of course, for any of the above studies to be useful, we need basic information on the location and population size of all salmon stocks in the Columbia River Basin.

10.3.2 Could Reductions or Alterations in Hatchery Releases Help Recover Threatened Snake River Salmonids?

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

1) Comparing the spawning and rearing index areas that have been exposed to significant numbers of hatchery fish to others that have been relatively free of hatchery influence.
2) Determining the ecological interactions and possible effects of hatchery fish releases on wild fish. 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 large concentrations of hatchery fish, and aggression) and suggest methods to minimize them. CRI researchers are currently exploring statistical relationships between magnitude and type of hatchery release and recruits per spawner data; unfortunately these analyses will have a problem separating cause and effect.
3) Producing a hatchery fish with characteristics more similar to those of wild fish may aid recovery of wild fish. However, a great deal of research is need to produce 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.
4) In many cases, conservation hatcheries 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.
5) Hatchery fish may improperly imprint during rearing or after release, potentially resulting in straying of returning adults and, thus, genetic introgression on wild stocks. Research should directly address a number of concerns over the potential effects of homing and 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.

10.3.3 Are Differential Delayed Transportation Mortality or Latent and Extra Mortality Caused by Factors that Indicate Dam Breaching Could Successfully Recover Snake River Salmonids?

The extent to which transported fish suffer differential delayed mortality is a crucial question because the answer strongly influences the possible advantage 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, however, require several years to obtain sufficient data because sample sizes of recaptured returning fish are typically low, the magnitude of differential 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.

One possible cause of extra mortality is that dams, by altering the range and quality of habitats which fry, parr, and smolts occupy, may also alter the ultimate fitness of these fish. One way to examine whether dams are an important source of extra mortality would involve comparing the size and fecundity of individuals completing their freshwater rearing in the hydropower corridor to those completing this life stage under more normative conditions. For example, fall chinook on the Hanford Reach could be compared with fall chinook from the Snake River to provide an estimate of the impact of the four lower Snake River dams on that species. Because there is a relationship between size and fecundity in fishes, comparing the length of individuals from both groups at the juvenile, outmigrating smolt, and returning adult stages would provide: 1) an estimate of the growth rates and survivorships of both groups during the freshwater rearing stage and 2) an estimate of the relationship between size at the juvenile stage and adult fecundity. This would allow an assessment of not only whether the hydropower corridor and more riverine areas provide different-quality rearing habitats, but also whether those differences translate to differences in adult fecundity. Determining whether the timing of spawning differs between the groups would also be important to monitor, since fecundity of older females is likely to be greater due to their greater size. This type of analysis can provide insight into more subtle, but potentially important effects of dams on salmonid populations that comparisons of survivorship alone cannot yield.

10.3.4 Could Management of Predators Yield Substantial Benefits for Threatened Salmonids?

Predators have major impacts on salmonids throughout their life cycle. Bass and other exotic predator eat salmonids in reservoirs, Caspian terns consume smolts at the mouth of the Columbia River, and marine predators (marine mammals and fish) are a major source of mortality as well. Two significant questions are:

1) What is the impact of different predators in terms of the percentage of salmonids eaten? If that were known, their impact on annual population growth would be straightforward to calculate.
2) What are the management options for reducing the impact of predators on salmon populations that are at risk?

These questions require research that involves multiple species and is less salmonid-centric than has been typical in the past. Importantly, predation is tied up with hatcheries, habitat, harvest, and hydropower – because all of these "H-factors" can influence the type of predators present, the numbers of predators present, and the behavior or feeding efficiency of predators.

10.3.5 How do Changing Ocean Conditions Affect Chances for Successful Recovery of Snake River Salmonids?

CRI analyses suggest that survival in the ocean is a key life history stage. Unfortunately, ocean conditions are little more than a “black box” for all salmonids, and there is a need for long-term research focused on the relationship between ocean conditions and salmonid population dynamics. This research will not help inform decisions over the next few years, but could help place population fluctuations in a broader context over the long term, so management actions might 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 percent), 1978 to 1990 (3.3 percent) and 1991 to 1995 (0.5 percent). In 1991 and later years, average survival declined to less than one-fifth the rate evident during the 1978 to 1990 period, and only one-tenth that observed prior to 1977. The magnitude of these changes is more striking when considered that for these coho stocks, there are no potential effects of extra or delayed mortality attributable to dams. Given such dramatic changes in SARs (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.

10.4 Conclusions Regarding Critical Uncertainties

Clearly, there are important uncertainties with substantial consequences for decisions about alternative management actions. It is equally clear that research can help resolve some of these uncertainties. However, research involves delay, and delay involves risk. The CRI extinction analyses provide a concrete measure of the risk of delaying action while learning more. These risks, which can be substantial, must be weighed against the value of identifying the feasibility of using particular management actions to achieve demographic improvements. Management itself represents an experiment, and there is certainly an opportunity to test the feasibility of options by careful monitoring and evaluation. Any management decisions that are made for the Snake River salmonids must be viewed as experiments from which we can learn information that might be applied to the many other populations of threatened and endangered salmonids throughout the West Coast. It must be emphasized that the extinction risks for several Snake River chinook salmon populations are so high that extinction is a real threat for this ESU. This argues for vigorous action.

11. Glossary

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: Acronym for bacterial kidney disease, 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 rate: 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, and subtracting out estimates of harvest and tributary harvest between the dams (see formula in Section 4.2.2).

CRiSP: Acronym for Columbia River Salmon Passage, 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-value: 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 no different between transported and nontransported 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 nontransported fish).

Extra mortality: 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: Fish Leaving Under Several Hypotheses (FLUSH) is 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 3.4.3.2). The current ocean regime, and a shift on the timescale of years or decades, may affect the likelihood of recovery under any 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 [1998b] for a brief comparison).

Recovery: The process by which the ecosystem is restored so that 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 not meeting recovery and survival criteria, an action that succeeds in satisfying performance criteria over the widest range of assumptions.

Survival: The persistence of the species 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.

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Annex A

PATH Snake River Spring/Summer Chinook Models

ANNEX A

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

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

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

$\lambda_{n,r} = \exp[-m_r - \ln(T_r)]$ 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 **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 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 **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 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 **input** 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:

$\alpha_{i,j} = \alpha_n$ - [average α_n]- $\ln(D_t P_t + 1 - P_t)$ + [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 **estimated** in the retrospective model from other terms in the model and from the following additional values, which are **input** 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.

$STEP_j$ 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 **<u>estimated</u>** in the model are:

c_1 , c_2 = estimated coefficients

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

Prospective Implementation

In the prospective Alpha model, the $\alpha_{t,\ j}$ 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 **<u>input</u>** 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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Annex B

Downstream Migrant Juvenile Salmonid Survival Estimates through the Snake and Columbia River Hydrosystem

1966 to 1980 and 1993 to 1998

ANNEX B: Downstream migrant juvenile salmonid survival estimates through the Snake and Columbia River hydrosystem, 1966 to 1980 and 1993 to 1998

NMFS scientists have estimated survival probabilities for juvenile salmonids migrating through sections of the Snake and Columbia Rivers during two periods since the mid-1960s. From 1966 to 1980, fish were mass branded and used to estimate populations at dams. Comparisons of populations at upstream and downstream dams were used to estimate survival. No estimates were made between 1981 and 1992. From 1993 to present, survival estimates were made from the detection records of PIT-tagged fish. During each year in each period, the survival research was conducted (survival estimates derived for) a subsection of the entire hydrosystem in place at the time. This paper summarizes the sections of river in which survival research was conducted and the estimates obtained (sections labeled AEstimation@), and describes methods used to extrapolate from the available reach estimates a calculation of the estimated survival probability through the entire hydrosystem for each year (sections labeled AExpansion@)

Period 1. 1966 -1980

Survival Estimation

Raymond (1979) provided survival estimates over much of the river reach that juvenile chinook salmon and steelhead migrated in the Snake and lower Columbia Rivers for the period 1966 through 1975. During these years, survival was estimated from Ice Harbor Dam on the Snake River to The Dalles Dam on the lower Columbia River. From 1966 through 1968, Ice Harbor Dam was the uppermost dam on the lower Snake River. As dams were completed above Ice Harbor Dam, survival was estimated from the uppermost dam (Lower Monumental Dam in 1969, Little Goose Dam 1970-74, and Lower Granite Dam in 1975) to Ice Harbor Dam. From 1969 through 1975, the product of survival estimates from the upper dam to Ice Harbor Dam and from Ice Harbor Dam to The Dalles Dam provided an overall estimate of survival for the reach between the upper dam on the Snake River to The Dalles Dam.

In addition to the NMFS studies reported by Raymond (1979), NMFS conducted studies using the same methods to estimate survival from Lower Granite Dam to John Day Dam from 1976 to 1980. The results were not published but were discussed extensively in PATH. The Hydropower Workgroup agreed on values to use for spring/summer chinook salmon for all years between 1966 and 1980 from the upper dam on the Snake River to lowermost dam on the Columbia River where studies were conducted. The workgroup determined that upward adjustment of about 3% to account for transportation from Little Goose Dam was required for survival estimates for chinook salmon reported in NMFS annual reports for 1978 and 1979. While the workgroup did not discuss estimates for steelhead, the same adjustment appears reasonable for steelhead for those 2 years.

The NMFS used different methods to estimate (1) survival from the upper Snake River dam to Ice Harbor Dam and (2) survival from Ice Harbor Dam to the lower river dam. Between the upper Snake River dam and Ice Harbor Dam, an estimate of the total seasonal population of fish that arrived at the upper dam was divided into an estimate of the population that arrived at Ice Harbor Dam. The quotient was an estimate of the proportion of fish that survived from the upper to the lower dam. This estimate included passage through the upper dam to arrival at, but not through Ice Harbor Dam. The first step for estimation of the total population that arrived at each dam was to estimate collection efficiency at each dam. Then the daily number of fish collected at each dam was expanded by the estimated collection efficiency to estimate the total daily number of fish that passed the dam. The daily estimates were summed for the season. The daily collection efficiency at a dam varied by the proportion of flow that passed through the powerhouse and spillway. It also depended somewhat on the level of smoltification of the cohort of fish that passed each day.

It was not possible to use the method of comparing the estimated total number of fish passing two dams for the reach including the lower river, because the population of fish that arrived at John Day Dam or The Dalles Dam included fish from the upper Columbia River and lower river tributaries. Instead, marked fish were released at Ice Harbor Dam (or at McNary Dam in some years) and the daily numbers of marked fish recovered at John Day Dam or The Dalles Dam were expanded by collection efficiency estimates to estimate the total number of marked fish that arrived at the dam. This estimate was divided by the number of marked fish released at Ice Harbor Dam to calculate the survival estimate over the reach. Marked fish were released and recovered throughout the migration season. Raymond (1979) provides details on the NMFS methodology and means used to ensure, to the extent possible, that the estimates were unbiased. If handling techniques used to capture, mark, release, and recover fish at each dam caused equal mortalities to test fish used for capture-efficiency estimates, then a comparison of the population estimates at two dams was considered reasonable. However, for the lower river survival estimates, it was unknown if the effect of marking and handling on fish that migrated to John Day Dam or The Dalles Dam was the same as on fish released just upstream of the dam to estimate collection efficiency. If higher mortality occurred between marking and recapture of fish released at Ice Harbor Dam compared to those released just upstream of John Day Dam or The Dalles Dam, then survival estimates in the lower river based on marked fish were likely lower than the survival of the population at large.

Estimated survival from the upper dam on the Snake River to either The Dalles Dam or John Day Dam (depending on the year as outlined above) is reported in Table 1. The number of projects (one Aproject@ equals one reservoir and one dam) is also indicated.

Expansion of Estimates to Entire Hydrosystem

An estimate of survival through the entire hydropower system (i.e., for river reaches in which survival research was not conducted) requires extrapolation of the reach survival estimates in Table 1. The estimates of survival in the Snake River for spring/summer chinook salmon and steelhead from 1966 through 1980 did not include the reservoir upstream of the uppermost dam, though it did include passage through the dam. The reach survival estimates in Table 1 also did not include the lower dam on the Columbia River at which fish were collected or the reservoirs and dam(s) downstream of the collection point.

Thus, for years in which The Dalles Dam was the lower collection point (1966-1975), the estimate did not include the upper Snake River reservoir, The Dalles Dam, Bonneville Dam reservoir, and Bonneville Dam. This is the equivalent of two projects. For years in which John Day Dam was the lower collection point (1976-1980), the estimate did not include the upper Snake River reservoir, John Day Dam, and The Dalles and Bonneville Dam projects; i.e., three projects.

To extrapolate the estimates from the survival research, we assumed that the per-project survival probability for the river sections outside the research section was the same as that estimated within the Ice Harbor-to-downstream dam reach. Between 1966 and 1975, survival was estimated between Ice Harbor Dam and The Dalles Dam. Fish released at Ice Harbor Dam passed through Ice Harbor, McNary, and John Day (1968-1975) dams, and through the McNary, John Day (1971-1975), and The Dalles Dam reservoirs. Thus, the overall survival estimates for that reach in 1966 and 1967 were taken to the 2 power to derive per-project survival estimates, and for 1968-1975 the estimate was taken to the 1/3 power. For 1976, 1978, and 1979, the survival estimate from Ice Harbor Dam to John Day Dam was taken to the 2 power, and for 1980 the survival estimate from Ice Harbor Dam to McNary Dam was used as the per-project survival outside the research reach. For 1977, survival in the lower reach was not estimated separately from the Snake River; the estimate from Lower Granite Dam to John Day Dam was taken to the 1/5 power to calculate per-project survival.

Thus, the overall system survival estimate (Table 2) was computed by multiplying estimated survival from the upper Snake River Dam to the lower river dam (Table Y) by the extrapolated probability estimated for projects outside the research reach.

Section 2. 1993-1998

Survival Estimation

From 1993 through 1998, NMFS has estimated survival of juvenile migrant salmonids using electronic PIT-tags and statistical methods for release-recapture data. Estimates for spring/summer chinook salmon in 1993 and 1994 and for steelhead in 1994 were restricted to the Snake River, beginning near the head of Lower Granite Dam reservoir and ending at Little Goose Dam (chinook salmon in 1993) or Lower Monumental Dam (both species in 1994). From 1995 through 1998, survival was estimated for both species to McNary Dam on the Columbia River, though the starting point for estimates for both species was moved downstream to the tailrace of Lower Granite Dam. Survival estimates from PIT-tag data were reported in annual contract reports for 1993 through 1996, and annual reports for 1997 and 1998 are in preparation. Each annual report has included annual average survival estimates. Averages have been weighted by the inverse of the respective estimated variances of the individual survival estimates. In addition, PATH has calculated averages weighted by inverse variances and passage indices jointly.

Recent analyses have advanced and refined the NMFS estimates of survival probabilities from PIT-tagged juvenile salmonids. This document presents the results of these recent calculations, giving yearly average estimates of survival probabilities for migration years 1993 through 1998. Estimates for 1993 through 1996 are intended to supersede average estimates previously published in our annual reports.

Table 4.7.1-5 in the PATH Final Report for FY 1998 included average survival estimates for 1994-1996 and preliminary information for 1997. The estimates reported here for those years differ from those in the PATH report for four reasons: (1) respective inverse estimated relative variance of each individual estimate was used for the weighted average, rather than the inverse estimated variance (see below); (2) daily Lower Granite Dam passage index was not used to weight individual estimates (also see below); (3) hatchery and wild fish were pooled in the recent analyses; and (4) the PATH report erroneously lists the number of projects for the Lower Granite Dam-to-McNary Dam estimates as 4.5, rather than the correct 4.0

Three differences from, and improvements over the NMFS= previous analyses are:

(1) Using information from PIT tags detected by a PIT-trawl below Bonneville Dam and those recovered from bird colonies on Rice Island, we have estimated survival probabilities for migrating steelhead in 1997 and 1998 from McNary Dam tailrace to Bonneville Dam tailrace directly, using the Single-Release Model. For chinook salmon, insufficient numbers of fish were detected and PIT tags recovered to estimate survival. However, we believe that approximate estimates for chinook salmon are inferrable from the estimates of steelhead survival.

(2) For weighted averages of multiple survival estimates within a season, we recognized a shortcoming of the use of inverse variance for weights. The estimated variance of a survival estimate from the SR Model is partly a function of the square of the survival estimate itself. Thus, if two estimates are based on the same amount of information (i.e., same number of detections contributing to a survival estimate), the lower survival estimate will have a smaller variance, and hence a larger weight in the weighted average if inverse variance is used. This problem has the greatest effect for

reaches where detection data are sparse (e.g., for estimates of survival to McNary Dam for several years of PIT-tag data). In such cases, survival estimates are more variable, and the lower estimates have disproportionately large influence on the inverse-variance-weighted-average, causing underestimation of the true mean. More appropriate weights are provided by the inverse of the respective relative variances[a], which weight the estimates essentially by the amount of data that contributed to them, and remove the influence of the survival estimates themselves.

Burnham et al (1987) express the estimated variance of the survival estimate $\hat{S}_i$ as:

$$\hat{var}(\hat{S}_i) = (\hat{S}_i)^2 \bullet f(r_i, R_i, r_{i+1}, R_{i+1}, A_{i+1}, T_{i+1}, \hat{P}_{i+1})$$

where $\hat{P}_{i+1}$ is an estimated detection probability and the rest of the quantities in the function are statistics based on counts of fish with specific detection records. Relative variance is defined as the variance divided by the square of the estimate. In the case of survival estimates from the Single-Release Model, then, the relative variance is a function of a detection probability and counting statistics, and is not influenced by the survival estimate itself. Thus, the weighted average of a series of estimates is given by:

$$\frac{\sum_{i=1}^{I} w_i \hat{S}_i}{\sum_{i=1}^{I} w_i}$$

where $w_i = \frac{(\hat{S}_i)^2}{\hat{var}(\hat{S}_i)}$.

Note: the weighted averages listed in Table 4.7.1-5 of the FY 98 PATH report are weighted jointly by both inverse variance and Lower Granite Dam passage index. The formula used can be expressed as:

$$w_i = \frac{Z_i \bullet \hat{var}(\hat{S}_i)^{-1}}{\sum(Z_i \bullet \hat{var}(\hat{S}_i)^{-1})}$$

where Z_i is the daily passage index normalized so that $\sum Z_i = 1$. Because the passage index component is normalized while the inverse variance component is not, the inverse

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[a] Relative variance is equal to the variance divided by the square of the survival estimate; i.e., the square of the coefficient of variation.

variance has much more influence on the weighted average than does the passage index. Therefore, this method has the same flaw as the inverse variance by itself.

(3) We also introduced another, more minor, adjustment to the method to correct a related source of bias (usually causing underestimates). In situations where some release groups of PIT-tagged fish are small (e.g., “adventitious” daily release groups from Lower Granite Dam made up of all fish PIT-tagged above Lower Granite Dam that were known to have passed on a particular day), it is not always possible to estimate survival through the longest reaches for the smallest release groups. In past analyses of adventitious groups, we have simply omitted daily release groups for which survival estimates were not possible. However, we have recognized that the omission of such groups causes bias in the following way: to estimate survival in the lower reaches of a survival study, a group must have a sufficient number of detections at the lower dams. For small groups which may have few fish remaining in the river in the lower reaches, there is an element of stochasticity in determining whether sufficient detections will occur to estimate survival. When there are sufficient detections, a high estimated detection probability usual results. When there are not enough detections, it is not possible to estimate either detection or survival probabilities. Because detection and survival probability estimates are negatively correlated in the statistical model, especially when groups are small, this means that with small groups, survival estimates are possible only when observed detection probabilities are high. Hence, the typical result is that either a low survival estimate is calculated, or no survival estimate is calculated at all. Instead of omitting a daily group where no survival estimate is possible, we have determined that a better approach is to pool that group with adjacent days until detections for the pooled group are sufficient to estimate survival. Survival estimates obtained in this way are generally higher on average than those based only on individual daily groups for which survival estimates were possible.

The reaches where survival estimates for PIT-tagged migrants were calculated start either at or near the head of Lower Granite Reservoir, or at the tailrace of Lower Granite Dam. NMFS purse-seined, PIT-tagged and released hatchery spring/summer chinook salmon in Lower Granite Reservoir from 1993 through 1995. Hatchery steelhead were tagged and released in the reservoir from 1994 through 1996. For both species, survival was estimated downstream from Lower Granite Dam from 1994 through 1998 by combining PIT-tagged fish into Aadventitious release groups@ composed of all tagged fish of the species known to have left Lower Granite Dam on the same day. These groups combined both wild and hatchery fish, and included both fish that were tagged at Lower Granite Dam then released into the tailrace and those that were tagged above Lower Granite Dam and detected and returned to the tailrace on the particular day.

For both starting points, survival probabilities were estimated downstream from the release point using the Single-Release Model to analyze records of PIT-tag detections for individual tagged fish. As the number of dams equipped with PIT-tag detectors and mechanisms for returning detected fish to the tailrace of the dam increased, the reach over which survival was estimated was extended downstream. In 1993, survival was estimated only from Lower Granite Reservoir to the tailrace of Little Goose Dam (and only for chinook salmon). In 1994, the lower limit of the reach was the tailrace of Lower Monumental Dam. Beginning in 1995, the lower limit was McNary Dam tailrace for most analyses. Inclusion of detection sites below Bonneville Dam in 1997 and 1998 allowed survival estimation from the tailrace of McNary Dam to the tailrace of Bonneville Dam for steelhead (detections of chinook salmon below Bonneville are too sparse for estimation).

In 1993, 7 groups of PIT-tagged spring/summer chinook salmon were released in Lower Granite Reservoir and their survival was estimated to Little Goose Dam. The weighted average (inverse relative variance) estimated survival was 0.75 (Table 3). For chinook salmon in 1994 and 1995 and steelhead in 1994-1996, survival over the largest reach possible was estimated as the product of two estimates (1) weighted average estimated survival to Lower Granite Dam tailrace for groups released in Lower Granite Reservoir, and (2) weighted average estimated survival from Lower Granite Dam tailrace to Lower Monumental or McNary Dam tailrace for daily groups of PIT-tagged fish leaving Lower Granite Dam (Table 3). For chinook salmon in 1996-1998 and steelhead in 1997 and 1998, there were no reservoir releases. The Aresearch reach@ for those years was Lower Granite Dam tailrace to the tailrace of the farthest downstream dam possible. For steelhead in 1997 and 1998, the estimate for McNary Dam to Bonneville Dam (Table 3) is the weighted average survival estimate for weekly adventitious groups leaving McNary Dam.

Expansion of Estimates to Entire Hydrosystem

Survival estimates in the Aresearch reach@ were expanded to estimate the overall hydrosystem survival probability (head of Lower Granite Reservoir to tailrace of Bonneville Dam) by applying the estimated per-project survival from the research reach to the projects for which survival was not estimated directly (Table 4). The projects (reservoir/dam combined) to which the per-project survival estimate was extrapolated included Lower Granite Dam project for some species in some years, and from the tailrace of the lower dam to the tailrace of Bonneville Dam for all species in all years except for steelhead in 1997 and 1998. For steelhead in 1997 and 1998, survival through Lower Granite Dam reservoir and dam was extrapolated from the PIT-tag estimated survival probability from Lower Granite Dam to McNary Dam.

Table 1. Reach survival estimates from the upper dam on the Snake River (Ice Harbor Dam 1966-68, Lower Monumental Dam 1969, Little Goose Dam 1970-74, and Lower Granite Dam 1975-80) to a lower dam on the Columbia River (either The Dalles Dam through 1975 or John Day Dam through 1980). Abbreviations: lgr-Lower Granite Dam; lgo-Little Goose Dam; lmo-Lower Monumental Dam; ihr-Ice Harbor Dam; mcn-McNary Dam; jda-John Day Dam; tda-The Dalles Dam.

	Survival from upper Snake River dam to Ice Harbor Dam			**X**	**Survival from Ice Harbor Dam to lower river dam**			**=**	**Survival from upper Snake River dam to lower river dam**		
Year	**Reach (no. projects)**	**Chinook salmon**	**Steelhead**		**Reach (no. projects)**	**Chinook salmon**	**Steelhead**		**Reach (no. projects)**	**Chinook salmon**	**Steelhead**
After Raymond (1979)											
1966	Ice Harbor was upper Snake River dam				ihr-tda (2)	0.63	0.75		ihr-tda (2)	0.63	0.75
1967					ihr-tda (2)	0.64	0.57		ihr-tda (2)	0.64	0.57
1968					ihr-tda (3)	0.62	0.60		ihr-tda (3)	0.62	0.60
1969	lmo-ihr (1)	0.75	0.85		ihr-tda (3)	0.62	0.42		lmo-tda (4)	0.47	0.36
1970	lgo-ihr (2)	0.33	0.80		ihr-tda (3)	0.67	0.48		lgo-tda (5)	0.22	0.38
1971	lgo-ihr (2)	0.48	0.80		ihr-tda (3)	0.59[a]	0.40[a]		lgo-tda (5)	0.28	0.32
1972	lgo-ihr (2)	0.39	0.60		ihr-tda (3)	0.42	0.33		lgo-tda (5)	0.16	0.20
1973	lgo-ihr (2)	0.12	0.27		ihr-tda (3)	0.42	0.15		lgo-tda (5)	0.05	0.04
1974	lgo-ihr (2)	0.50	0.78		ihr-tda (3)	0.71	0.25		lgo-tda (5)	0.36	0.20
1975	lgr-ihr (3)	0.36	0.74		ihr-tda (3)	0.69	0.55		lgr-tda (6)	0.25	0.41
From unpublished NMFS data											
1976	lgr-ihr (3)	0.63	0.72		ihr-jda (2)	0.48	0.50		lgr-jda (5)	0.30	0.36
1977	--------not estimated separately--------								lgr-jda (5)	0.03	0.02
1978	lgr-ihr (3)	0.69	0.71		ihr-jda (2)	0.64	0.42		lgr-jda (5)	0.47[b]	0.33[b]
1979	lgr-ihr (3)	0.43	0.14		ihr-jda (2)	0.72	0.46		lgr-jda (5)	0.34[b]	0.10[b]
1980	lgr-mcn (4)	0.49	0.41		mcn-jda (1)	0.74	0.50		lgr-jda (5)	0.36	0.21

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[a] No estimate of survival from Ice Harbor Dam to lower river dam is available for this year. The average estimate from 1968-1970 and 1972-1975 was used.

[b] Increased by 0.03-0.04 from annual report to adjust for transportation from Little Goose Dam.

Table 2. System survival estimates for 1966-1980 from the upper reservoir on the Snake River (Ice Harbor Dam 1966-68, Lower Monumental Dam 1969, Little Goose Dam 1970-74, and Lower Granite Dam 1975-80) to the tailrace of Bonneville Dam. Abbreviations: lgr-Lower Granite Dam; lgo-Little Goose Dam; lmo-Lower Monumental Dam; ihr-Ice Harbor Dam; mcn-McNary Dam; jda-John Day Dam; tda-The Dalles Dam; bon-Bonneville Dam.

	Survival from upper Snake River dam to lower river dam			**X**	**Extrapolated survival outside research reach**			**=**	**Overall system survival**		
Year	**Reach (# projects)**	**Chinook salmon**	**Steelhead**		**Reach (# projects)**	**Chinook salmon**	**Steelhead**		**Reach (# projects)**	**Chinook salmon**	**Steelhead**
	After Raymond (1979) (See Table 1)										
1966	ihr-tda (2)	0.63	0.75		2	0.63	0.75		ihr-bon (4)	0.40	0.56
1967	ihr-tda (2)	0.64	0.57		2	0.64	0.57		ihr-bon (4)	0.41	0.32
1968	ihr-tda (3)	0.62	0.60		2	0.73	0.71		ihr-bon (5)	0.45	0.43
1969	lmo-tda (4)	0.47	0.36		2	0.73	0.56		lmo-bon (6)	0.34	0.20
1970	lgo-tda (5)	0.22	0.38		2	0.77	0.61		lgo-bon (7)	0.17	0.24
1971	lgo-tda (5)	0.28	0.32		2	0.71	0.54		lgo-bon (7)	0.20	0.17
1972	lgo-tda (5)	0.16	0.20		2	0.56	0.48		lgo-bon (7)	0.09	0.09
1973	lgo-tda (5)	0.05	0.04		2	0.56	0.28		lgo-bon (7)	0.03	0.01
1974	lgo-tda (5)	0.36	0.20		2	0.80	0.40		lgo-bon (7)	0.28	0.08
1975	lgr-tda (6)	0.25	0.41		2	0.78	0.67		lgr-bon (8)	0.19	0.27
	From unpublished NMFS data										
1976	lgr-jda (5)	0.30	0.36		3	0.33	0.35		lgr-bon (8)	0.10	0.13
1977	lgr-jda (5)	0.03	0.02		3	0.12	0.10		lgr-bon (8)	0.04	0.02
1978	lgr-jda (5)	0.47	0.33		3	0.51	0.27		lgr-bon (8)	0.24	0.09
1979	lgr-jda (5)	0.34	0.10		3	0.61	0.31		lgr-bon (8)	0.21	0.03
1980	lgr-jda (5)	0.36	0.21		3	0.41	0.13		lgr-bon (8)	0.15	0.03

Table 3. Reach survival estimates from PIT-tag data, 1993-1998. Abbreviations: res-Lower Granite Dam reservoir; lgr-Lower Granite Dam; lgo-Little Goose Dam; lmo-Lower Monumental Dam; mcn-McNary Dam.

	Survival through Lower Granite Reservoir and Dam		**X**	**Survival from Lower Granite Dam tailrace to tailrace of lower dam**			**=**	**Survival from Lower Granite Reservoir to tailrace of lower dam**		
Year	**Chinook salmon**	**Steelhead**		**Reach (# projects)**	**Chinook salmon**	**Steelhead**		**Reach (# projects)**	**Chinook salmon**	**Steelhead**
1993	0.89	C		lgr-lgo (1)	0.84	C		res-lgo (2)	0.75	C
1994	0.92	0.90		lgr-lmo (2)	0.70	0.77		res-lmo (3)	0.64	0.69
1995	0.92	0.91		lgr-mcn (4)	0.72	0.74		res-mcn (5)	0.66	0.67
1996	C	0.94		lgr-mcn (4)	0.65	0.69		res-mcn (5)	C	0.65
1997	C	C		lgr-mcn (4)	0.65	0.73		lgr-bon (7)	C	0.47
				mcn-bon (3)	C	0.65				
1998	C	C		lgr-mcn (4)	0.77	0.65		lgr-bon (7)	C	0.50
				mcn-bon (3)	C	0.77				

Table 4. System survival estimates for 1993-1998 from Lower Granite Dam reservoir to the tailrace of Bonneville Dam. Abbreviations: res-Lower Granite Dam reservoir; lgr-Lower Granite Dam; lgo-Little Goose Dam; lmo-Lower Monumental Dam; mcn-McNary Dam; bon-Bonneville Dam.

	Research reach (# projects)		**Survival through research reach**		**X**	**Extrapolated survival outside research reach**			**=**	**Overall system survival**	
Year	**Chinook salmon**	**Steelhead**	**Chinook salmon**	**Steelhead**		**Reach (# projects)**	**Chinook salmon**	**Steelhead**		**Chinook salmon**	**Steelhead**
1993	res-lgo (2)	C	0.75	C		lgo-bon (6)	0.43	C		0.32	C
1994	res-lmo (3)	res-lmo (3)	0.64	0.69		lmo-bon (5)	0.48	0.54		0.31	0.38
1995	res-mcn (5)	res-mcn (5)	0.66	0.67		mcn-bon (3)	0.78	0.79		0.51	0.53
1996	lgr-mcn (4)	res-mcn (5)	0.65	0.65		res-lgr (1) mcn-bon (3)	0.90 0.72	C 0.77		0.42	0.50
1997	lgr-mcn (4)	lgr-bon (7)	0.65	0.47		res-lgr (1) mcn-bon (4)	0.90 0.72	0.92 C		0.43	0.44
1998	lgr-mcn (4)	lgr-bon (7)	0.77	0.50		res-lgr (1) mcn-bon (4)	0.94 0.82	0.90 C		0.59	0.45

Annex C

Rationale Behind NMFS Approach to Estimation of "D" from PIT-Tag Data

ANNEX C: Rationale Behind NMFS Approach to Estimation of "D" from PIT-Tag Data

While the concept of differential post-Bonneville survival for transported and inriver fish is general, the parameter 'D' has a specific meaning, given by the manner in which it is applied in the PATH life-cycle models. There, 'D' is defined as the ratio of two parameters: λ_T, the post-Bonneville survival for transported fish, and λ_C, the post-Bonneville survival for fish that arrive below Bonneville via in-river routes. In particular, the traditional "T:C" ratio of Lower Granite smolt-to Lower Granite adult return rates for the two groups can be expressed as the product of the ratio of juvenile survival from Lower Granite Dam to Bonneville Dam and the ratio of post-Bonneville Dam survival:

$$T{:}C = \frac{SAR_T}{SAR_C} = \frac{V_T \lambda_T}{V_C \lambda_C} = \frac{V_T}{V_C} D \cdot$$

Despite evidence that post-Bonneville survival for transported fish varies depending on the dam from which fish were transported (in particular, fish transported from McNary Dam appear to have lower return rates than those transported from Lower Granite or Little Goose Dam, as discussed below), the PATH life-cycle models assign the same value of λ_T, and hence *D*, to all transported fish, regardless of the dam from which they were transported. Thus, if post-Bonneville survival does vary depending on transport site, the PATH *D* is actually a weighted average of the differential mortality for the various transport sites included in a particular prospective scenario.

(In addition, the PATH models apply the same *D* value to all transported fish regardless of the date which they were released below Bonneville Dam. PIT-tag data from 1995 provide evidence of important seasonal variations in post-Bonneville survival of transported fish. More years of such data are needed).

Moreover, all previous PATH analyses (non PIT-tag) that attempted to estimate *D* were based on transport studies that transported fish from Lower Granite or Little Goose Dams. The resulting estimated *D* values have then been applied to all transported fish in the PATH models. In NMFS' analysis in the previous AFISH draft, our choice to use fish transported only from Lower Granite or Little Goose Dams was in part to be consistent with these previous analyses, and in part because most prospective scenarios involving transportation place heavy emphasis on collecting and transporting fish at the upper dams. The States and Tribes' (STFA) analysis is perhaps the first to attempt to estimate *D* from fish transported from all four transport dams (Schaller et al 1999).

When using data from PIT-tagged fish to estimate parameters for the PATH models, it is important to remember that those models are intended to represent the runs at large, and that PIT-tagged fish are not necessarily representative of nontagged fish in every regard. Especially important in the case of estimating *D* is the fact that the proportions of PIT-tagged fish that experience certain detection histories is vastly different from the proportions of nontagged fish. It was this realization that led to the use of "never detected" PIT-tagged fish as the most proper group to use to represent nontagged fish that remain in the river. PIT-tagged fish that entered collection systems in 1994-1996 were usually returned to the river, nontagged fish in collection systems were transported. (The situation changed beginning in 1997, when many PIT-tagged hatchery fish were purposefully transported from Lower Granite Dam for the Idaho Hatchery PIT-Tag Study). Thus, of fish that remained in the river and survived to Bonneville Dam, a much higher proportion of PIT-tagged fish experienced one or more bypass systems than did their nontagged counterparts.

The same care must be taken to define the group of transported PIT-tagged fish that is to represent transported nontagged fish to estimate *D* for the PATH models. Most PIT-tagged fish were returned to the river at Lower Granite and Little Goose dams. The result is that, comparing transported PIT-tagged and transported nontagged fish, a higher proportion of PIT-tagged fish were transported from lower dams than their nontagged counterparts. To say it another way, nontagged fish were transported the first time they were bypassed; more PIT-tagged fish were returned to the river and "vulnerable" to transportation at lower dams. Estimates of *D* based on PIT-tag data must account for this bias toward lower-river transport among PIT-tagged fish.

The bias was particularly strong in 1994, before McNary Dam was equipped with a slide-gate, so that all PIT-tagged fish bypassed there were transported. Of the total number of PIT-tagged wild yearling chinook salmon (Lower Granite-equivalents) transported in 1994, the proportions transported from Lower Granite, Little Goose, Lower Monumental, and McNary dams were 9%, 7%, 10%, and 75%, respectively (rounding accounts for the total of 101%). In contrast, we estimate roughly the following proportions among transported nontagged fish in 1994: 45%, 15%, 25%, 15%.

The STFA analysis adds together PIT-tagged fish transported from all sites and considers them representative of nontagged transported fish. We estimated return rates for wild PIT-tagged fish transported from Lower Granite, Little Goose, Lower Monumental, and McNary dams of 0.69%, 0.59%, 0.08%, and 0.02%, respectively. The STFA report notes that the choice of inclusion or exclusion of fish transported from Lower Monumental and McNary dams has the greatest influence on the estimate of *D*. This result is almost entirely due to the great difference in return rates for fish transported from various dams in 1994, and the failure of the STFA analysis to properly construct a PIT-tagged transport group representative of nontagged transported fish in that year. Because very few fish, tagged or nontagged, were transported from McNary Dam in 1995 or 1996, the effect is not nearly as big for those years.

Using the assumptions we used in the previous draft, the estimated *D* value was 1.24 for wild yearling chinook salmon in 1994, based only on fish transported from Lower Granite or Little Goose Dam. If we simply added together fish transported from all four transport sites, as was done by STFA, the estimate was drastically changed, to 0.24. However, this estimate was not a valid representation of the PATH-model parameter, because the PIT-tagged transported group was not representative of the run at large. To properly represent nontagged fish, the return rates from juveniles transported from the various dams must be weighted proportionally to nontagged fish transported from each dam (roughly 45%, 15%, 25%, 15%, as noted above).

When this was done, the estimated *D* value for wild chinook salmon in 1994 was 0.82. To make a useful contribution, STFA must redo their analysis, correctly handling fish transported from the lower dams. We suspect the previous NMFS results will not appear as "extreme."

The second most influential alternative in the STFA analysis was the method used to extrapolate empirical survival estimates from the Snake River to the stretch from McNary Dam to Bonneville Dam, where no empirical data could be collected in 1994-1996. NMFS assumed per-project survival was the same in the lower river as in the Snake, while STFA proposed extrapolation based on equal per-mile survival probabilities. Empirical estimates of McNary-to-Bonneville survival are now available for PIT-tagged steelhead in 1997, 1998, and 1999, and for PIT-tagged yearling chinook salmon in 1999. The following table compares each empirical estimate with values extrapolated by the two methods from estimated Lower Granite-to-McNary survival from the same year:

Species/Year	Empirical estimate survival MCN-BON	Per-project extrap.	Per-km extrap.
1997 steelhead	0.651	0.788	0.717
1998 steelhead	0.769	0.729	0.635
1999 steelhead	0.720	0.759	0.679
1999 chinook	0.715	0.839	0.782

For steelhead, per-km extrapolation was more accurate in 1997, per-project was more accurate in 1998, and there was virtually no difference in accuracy in 1999. Both extrapolations overestimated for chinook salmon in 1999; per-project more so. Available empirical data remain too sparse to resolve the question of proper extrapolation method for years before lower-river estimates were available. Perhaps the two methods bracket the reasonable range of possibilities.

The STFA report states that "more data are unlikely to perfect our understanding of 'D' or eliminate the uncertainty in the most influential assumptions." This statement does not follow from the conclusions presented in the STFA report itself and is easily refuted: the report notes that the two most influential assumptions on *D* estimates are (1) whether or not PIT-tagged fish transported from Lower Monumental or McNary dams are included in the “transport” group; and (2) the method used to extrapolate survival estimates to the McNary-to-Bonneville stretch. This document demonstrates that (1) is not really an uncertainty about assumptions, but about the proper way to use PIT-tag data to represent the relevant groups in the PATH life-cycle models. This document also shows how this “not likely resolvable” uncertainty is solved. Influential, “unresolvable” assumption (2), has also already been resolved by continued development of the PIT-tag detection system, so that extrapolation to the lower river is no longer necessary. For juvenile steelhead migrations beginning in 1997 and yearling chinook salmon migrations beginning in 1999, empirical data are the basis of the McNary-to-Bonneville survival estimate.

The PIT-tag system continues to develop, along with our understanding of post-Bonneville survival and how to investigate it with PIT-tag and other data. Ongoing direct experiments directed to resolve remaining uncertainties surrounding D are indeed the key to answering the age-old question “Does transportation work?”

Literature Cited

Schaller, H. et al. 1999. An analysis of differential delayed mortality (‘D’) experienced by stream-type chinook salmon of the Snake River: A response by State, Tribal and USFWS technical staff to the ‘D’ analyses and discussion in the Anadromous Fish Appendix to the U.S. Army Corps of Engineers’ Lower Snake River Juvenile Salmonid Migration Feasibility Study. Draft dated August 12, 1999.

Annex D

Annual Population Growth Rate and Risks, Assuming that Hatchery Fish Reproductive Success is 20 and 80 Percent that of Wild Fish

Table D-1. Needed incremental change from base period survival to achieve 5% risk of extinction in 24 years

		Lambda Calculated From 1980 to Most Recent Completed Year							
		20% Historical Effectiveness of Hatchery Spawners				80% Historical Effectiveness of Hatchery Spawners			
	Mean Gen. Time	Estimated Lambda	Lambda Needed to Meet Criterion	Necessary % Change in Lambda	Necessary % Change in Survival	Estimated Lambda	Lambda Needed to Meet Criterion	Necessary % Change in Lambda	Necessary % Change in Survival
Snake River Spring/Summer Chinook									
Aggregate ESU	4.73	0.91	0.91	0.00	0.00	0.82	0.82	0.00	0.00
Bear Valley/Elk Creeks	4.729	1.02	1.02	0.00	0.00	1.02	1.02	0.00	0.00
Imnaha River[1]	4.486	0.89	0.89	0.00	0.00	0.88	0.88	0.00	0.00
Johnson Creek	4.351	1.01	1.01	0.00	0.00	1.01	1.01	0.00	0.00
Marsh Creek	4.684	0.99	0.99	0.00	0.00	0.99	0.99	0.00	0.00
Minam River	4.178	0.98	0.98	0.00	0.00	0.93	0.93	0.00	0.00
Poverty Flats	4.221	1.00	1.00	0.00	0.00	0.99	0.99	0.00	0.00
Sulphur Creek	4.610	1.04	1.04	0.00	0.00	1.04	1.04	0.00	0.00
1 50%, rather than 20%, effectiveness of hatchery-origin natural spawners was applied to the Imnaha index stock.									
Alturas Lake Ck	4.465	0.75				0.75			
American R	4.465	0.91				0.91			
Big Sheep Ck	4.465	0.88				0.85			
Beaver Cr	4.465	0.95				0.95			
Bushy Fork	4.465	0.98				0.98			
Camas Cr	4.465	0.92				0.92			
Cape Horn Cr	4.465	1.05				1.05			
Catherine Ck	4.465	0.85				0.78			
Catherine Ck N Fk	4.465	0.92				0.92			
Catherine Ck S Fk	4.465	0.80				0.80			
Crooked Fork	4.465	1.00				1.00			
Grande Ronde R	4.465	0.84				0.77			
Knapp Cr	4.465	0.89				0.89			
Lake Cr	4.465	1.06				1.06			
Lemhi R	4.465	0.98				0.98			
Lookingglass Ck	4.465	0.79				0.72			
Loon Ck	4.465	1.00				1.00			
Lostine Ck	4.465	0.90				0.87			
Lower Salmon R	4.465	0.92				0.92			
Lower Valley Ck	4.465	0.92				0.92			
Moose Ck	4.465	0.94				0.94			
Newsome Ck	4.465	1.03				1.03			
Red R	4.465	0.91				0.91			
Salmon R E Fk	4.465	0.94				0.94			
Salmon R S Fk	4.465	1.06				1.06			
Secesh R	4.465	0.98				0.98			
Selway R	4.465	0.91				0.91			
Sheep Cr	4.465	0.80				0.80			
Upper Big Ck	4.465	0.97				0.97			
Upper Salmon R	4.465	0.90				0.90			
Upper Valley Ck	4.465	1.03				1.03			
Wallowa Ck	4.465	0.86				0.86			
Wenaha R	4.465	0.90				0.84			
Whitecap Ck	4.465	0.90				0.90			
Yankee Fork	4.465	0.88				0.88			
Yankee West Fk	4.465	0.99				0.99			
Snake River Fall Chinook									
Aggregate	4.137	0.92	0.92	0.00	0.00	0.87	0.87	0.00	0.00
Snake River Steelhead									
ESU Aggregate	5.168	0.83	0.83	0.00	0.00	0.72	0.72	0.00	0.00
A-Run Aggregate	5.040	0.85	0.85	0.00	0.00	0.74	0.74	0.00	0.00
B-Run Aggregate	6.490	0.84	0.84	0.00	0.00	0.74	0.74	0.00	0.00

A "Necessary % Change in Lambda" of, for example, 15.00 means that the median annual population growth rate ("Estimated Lambda") must be multiplied by 1.15 to meet the recovery criterion.

A "Necessary % Change in Survival" of, for example, 81.12 means that the average 1980-to-most-recent-year egg-to-adult survival rate rate, or any component life-stage survival rate, must be multiplied by 1.8112 to meet the recovery criterion.

Table D-2. Needed incremental change from base period survival to achieve 5% risk of extinction in 100 years

		Lambda Calculated From 1980 to Most Recent Completed Year							
		20% Historical Effectiveness of Hatchery Spawners				80% Historical Effectiveness of Hatchery Spawners			
	Mean Gen. Time	Estimated Lambda	Lambda Needed to Meet Criterion	Necessary % Change in Lambda	Necessary % Change in Survival	Estimated Lambda	Lambda Needed to Meet Criterion	Necessary % Change in Lambda	Necessary % Change in Survival
<u>Snake River Spring/Summer Chinook</u>									
Aggregate ESU	4.73	0.91	0.93	1.50	7.30	0.82	0.93	14.00	85.83
Bear Valley/Elk Creeks	4.729	1.02	1.02	0.00	0.00	1.02	1.02	0.00	0.00
Imnaha River[1]	4.486	0.89	0.96	7.50	38.32	0.88	0.96	9.50	50.24
Johnson Creek	4.351	1.01	1.01	0.00	0.00	1.01	1.01	0.00	0.00
Marsh Creek	4.684	0.99	1.02	3.00	14.85	0.99	1.02	3.00	14.85
Minam River	4.178	0.98	1.02	4.50	20.19	0.93	1.02	9.50	46.11
Poverty Flats	4.221	1.00	1.00	0.00	0.00	0.99	0.99	0.00	0.00
Sulphur Creek	4.610	1.04	1.11	7.00	36.60	1.04	1.11	7.00	36.60
1 50%, rather than 20%, effectiveness of hatchery-origin natural spawners was applied to the Imnaha index stock.									
Alturas Lake Ck	4.465	0.75				0.75			
American R	4.465	0.91				0.91			
Big Sheep Ck	4.465	0.88				0.85			
Beaver Cr	4.465	0.95				0.95			
Bushy Fork	4.465	0.98				0.98			
Camas Cr	4.465	0.92				0.92			
Cape Horn Cr	4.465	1.05				1.05			
Catherine Ck	4.465	0.85				0.78			
Catherine Ck N Fk	4.465	0.92				0.92			
Catherine Ck S Fk	4.465	0.80				0.80			
Crooked Fork	4.465	1.00				1.00			
Grande Ronde R	4.465	0.84				0.77			
Knapp Cr	4.465	0.89				0.89			
Lake Cr	4.465	1.06				1.06			
Lemhi R	4.465	0.98				0.98			
Lookingglass Ck	4.465	0.79				0.72			
Loon Ck	4.465	1.00				1.00			
Lostine Ck	4.465	0.90				0.87			
Lower Salmon R	4.465	0.92				0.92			
Lower Valley Ck	4.465	0.92				0.92			
Moose Ck	4.465	0.94				0.94			
Newsome Ck	4.465	1.03				1.03			
Red R	4.465	0.91				0.91			
Salmon R E Fk	4.465	0.94				0.94			
Salmon R S Fk	4.465	1.06				1.06			
Secesh R	4.465	0.98				0.98			
Selway R	4.465	0.91				0.91			
Sheep Cr	4.465	0.80				0.80			
Upper Big Ck	4.465	0.97				0.97			
Upper Salmon R	4.465	0.90				0.90			
Upper Valley Ck	4.465	1.03				1.03			
Wallowa Ck	4.465	0.86				0.86			
Wenaha R	4.465	0.90				0.84			
Whitecap Ck	4.465	0.90				0.90			
Yankee Fork	4.465	0.88				0.88			
Yankee West Fk	4.465	0.99				0.99			
<u>Snake River Fall Chinook</u>									
Aggregate	4.137	0.92	0.96	5.00	22.37	0.87	0.95	8.50	40.15
<u>Snake River Steelhead</u>									
ESU Aggregate	5.168	0.83	0.90	8.00	48.84	0.72	0.89	23.00	191.49
A-Run Aggregate	5.040	0.85	0.90	5.50	30.98	0.74	0.89	20.00	150.65
B-Run Aggregate	6.490	0.84	0.93	11.00	96.85	0.74	0.92	23.50	293.48

A "Necessary % Change in Lambda" of, for example, 1.50 means that the median annual population growth rate ("Estimated Lambda") must be multiplied by 1.015 to meet the recovery criterion.

A "Necessary % Change in Survival" of, for example, 7.30 means that the average 1980-to-most-recent-year egg-to-adult survival rate rate, or any component life-stage survival rate, must be multiplied by 1.073 to meet the recovery criterion.

Annex D.xls, Table D-2, 6/13/01