Application of decision analysis to evaluate recovery actions for threatened Snake River spring and summer chinook salmon (*Oncorhynchus tshawytscha*)

Calvin N. Peters and David R. Marmorek

Abstract: There is uncertainty about the importance of various factors in explaining declines of chinook salmon (*Oncorhynchus tshawytscha*) populations in the Snake River basin of Oregon and Idaho. This uncertainty has prevented implementation of long-term recovery actions for these stocks. We used simulation models and decision analysis to evaluate three management actions for seven index stocks of Snake River spring and summer chinook salmon: (*i*) continue current operation of the Columbia River hydropower system, (*ii*) maximize transportation of smolts, and (*iii*) natural river drawdown (breaching) of four Snake River dams. Decision analysis provided a useful approach for including multiple hypotheses about population responses to environmental and anthropogenic factors, systematically assessing the importance of alternative hypotheses, and identifying risk-averse recovery strategies that meet survival and recovery goals over a wide range of uncertainties. We found that the most influential uncertainties were related to hypothesized causes of estuary and ocean mortality. Current monitoring provides limited information on survival in this life stage; carefully designed management experiments are more likely to generate useful information. Given that these uncertainties exist, drawdown was the most risk-averse action, meeting long-term survival and recovery goals over a wider range of assumptions than the other actions.

Résumé : Il existe de l'incertitude au sujet de l'importance des divers facteurs explicatifs du déclin des populations de Saumons quinnat (*Oncorhynchus tshawytscha*) dans le bassin versant de la rivière Snake en Oregon et en Idaho. Cette incertitude a empêché la mise en oeuvre de stratégies de récupération à long terme de ces stocks. Des modèles de simulation et des analyses décisionnelles ont permis d'évaluer trois stratégies de gestion pour sept stocks indicatifs de saumons de printemps et d'été de la rivière Snake : (*i*) ou bien la poursuite de l'opération actuelle du système hydroélectrique du fleuve Columbia, (*ii*) ou alors la maximisation du transport des saumoneaux, (*iii*) ou enfin la baisse des eaux de la rivière à leur niveau naturel par ébrèchement de quatre barrages de la rivière Snake. L'analyse décisionnelle est une méthode utile pour incorporer des hypothèses multiples au sujet des réactions des populations aux facteurs environnementaux et anthropiques, pour évaluer systématiquement l'importance des hypothèses de rechange et pour identifier les stratégies de récupération qui évitent les risques tout en atteignant les objectifs de survie et de récupération sur une gamme étendue d'incertitudes. Les incertitudes les plus sérieuses concernent les causes présumées de la mortalité dans l'estuaire et l'océan. Le réseau de surveillance actuel ne fournit que peu de renseignements sur la survie à cette étape du cycle biologique; des expériences de gestion planifiées avec soin seraient plus susceptibles de générer des informations utiles. Étant donné l'existence de ces incertitudes, la baisse des niveaux des réservoirs est la stratégie qui pose le moins de risques et qui permet d'atteindre les objectifs de survie et de récupération à long terme sur une gamme plus étendue de pré-suppositions que les autres stratégies.

[Traduit par la Rédaction]

Introduction

Chinook salmon (*Oncorhynchus tshawytscha*) and steelhead (*Oncorhynchus mykiss*) populations in the Snake River sub-basin of the Columbia River have declined over the last 100 years and are now listed as "threatened" under the United States Endangered Species Act (NMFS 1995). The decline in abundance has been particularly dramatic since the mid-1970s (Schaller et al. 1999), a period that saw the completion of the eight dams of the Federal Columbia River Power System (FCRPS), the start of a large-scale transportation program in which smolts are collected at upper dams and transported by truck or barge to below the lowermost dam, an increase in hatchery smolt production, the degrada-

Received August 16, 2000. Accepted September 26, 2001. Published on the NRC Research Press Web site at http://cjfas.nrc.ca on December 18, 2001.
J15924

Calvin N. Peters.[1] ESSA Technologies Ltd., 4612 Briggs Rd., Vernon, BC V1B 3J4, Canada.
D.R. Marmorek. ESSA Technologies Ltd., 300–1765 W. 8th Avenue, Vancouver, BC V6J 5C6, Canada.

[1]Corresponding author (e-mail: cpeters@essa.com).

 DOI: 10.1139/cjfas-58-12-2431

tion of freshwater spawning and rearing habitat in some natal streams, and an abrupt shift in climate conditions in the Northeast Pacific Ocean (Beamish et al. 1999).

Disagreements among scientists and policy makers about the relative contribution of these factors to declines in Snake River salmon populations led to the development of several alternative modeling frameworks for estimating the impacts of historical factors and predicting the effects of proposed management actions. Because each model differed in its underlying assumptions about which historical factors were most important, each suggested different optimal recovery strategies. After several years of debates and conflicting management advice, the models were subjected to technical review and comparison by a panel of experts, which recommended that modelers and policy makers work together to evaluate the underlying hypotheses that were ultimately responsible for differences among models (Barnthouse 1993). This focus on collaboratively evaluating model hypotheses was reiterated in a 1994 judicial ruling on a salmon-related lawsuit (IDFG v. NMFS, 850F, Supp. 886, D.Or. 1994) and in the National Marine Fisheries Service's 1995 Biological Opinion (a document that assesses the current status of stocks and prescribes specific recovery actions; NMFS 1995). These documents led to the formation of the Plan for Analyzing and Testing Hypotheses (PATH), a multiagency research program to identify and resolve biological uncertainties surrounding recovery of threatened Snake River chinook salmon and steelhead. PATH operated from September 1995 to May 2000. Strengths and weaknesses of the PATH process are described in Marmorek and Peters (2001).

PATH "retrospective" analyses of historical data attempted to identify major spatial and temporal patterns in abundance and productivity of Snake River salmon stocks over the last 40 years and to determine the relative influence of habitat, harvest, hatchery, hydrosystem, and climatic factors on these patterns (Marmorek et al. 1996*a*; Schaller et al. 1999; Botsford and Paulsen 2000). These analyses resolved some differences in underlying assumptions about what has affected stocks historically (Marmorek et al. 1996*b*; Barnthouse et al. 2000; Petrosky et al. 2001). However, significant uncertainties about the likely effects of future management actions on Snake River chinook stocks persist because of limitations in data on past and present conditions, different interpretations of existing data, and uncertainty about future conditions (e.g., trends in future climate conditions or the effects of new actions that have not been tried before). These uncertainties make it difficult to project the outcomes of alternative recovery policies and thus to identify the best policies to implement.

PATH used formal decision analysis (Clemen 1996; Peterman and Anderson 1999) to evaluate recovery strategies for Snake River ocean-type fall-run chinook salmon (Peters et al. 2001) and for seven index populations of stream-type spring- and summer-run chinook that spawn in tributaries to the Snake River upstream of Lower Granite Dam (this study). These seven index stocks were assumed to be representative of the entire Snake River spring and summer chinook Evolutionarily Significant Unit (a group of stocks that are considered to be genetically distinct under the Endangered Species Act). The objectives of the PATH decision analysis were to determine the relative influence of various sources of uncertainty on the outcomes of alternative actions and to assess the performance of alternative recovery actions across uncertainties.

Methods

Overview

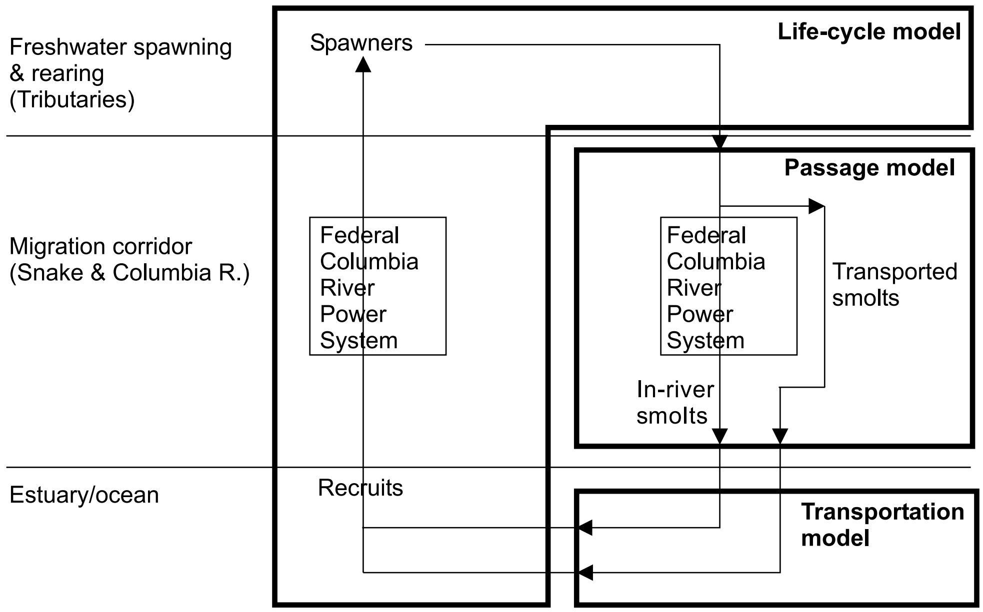
Decision analysis is a systematic approach to uncertainties in decision making using models to project outcomes of alternative actions. These outcomes (or performance measures) act as criteria for comparing and ranking the actions. Many possible outcomes are generated for each action, each resulting from a unique combination of alternative hypotheses about uncertain states of nature (for example, historical data may suggest that a range of spawner-to-recruit survival rates are possible). Such an approach has several benefits over approaches in which uncertainties are ignored or treated in an ad hoc manner. First, this approach can lead to more risk-averse management decisions in the long run because it allows decision makers to select "robust" actions that perform well under a broad range of assumptions. Implementing actions that are robust to uncertainties minimizes risks that the selected action will not have its intended effect, because such actions have a much higher chance of producing a favorable outcome even if the underlying hypotheses turn out to be wrong. Second, including uncertainties allows risk-averse actions to be identified and implemented before these uncertainties are fully resolved. This avoids the common situation in which uncertainties in the predicted effects of management actions discourage decision makers from changing management policies from the status quo (Peterman and Anderson 1999). Avoiding such inaction is particularly important for endangered species, where recovery actions generally need to be taken long before uncertainties can be resolved to the satisfaction of decision makers. Third, explicitly incorporating alternative hypotheses about key uncertainties allows scientists and agencies with alternative interpretations of existing data to participate in collaborative analytical processes.

There were seven components of the PATH decision analysis: alternative actions, performance measures, models, uncertainties, probabilities on alternative hypotheses, an overall decision analysis framework, and sensitivity analyses. Each component is described below.

Alternative actions to be evaluated

The PATH decision analysis focused primarily on three recovery actions related to the configuration and operation of eight Federally owned and operated dams on the Columbia and lower Snake rivers. Action A1 represented current operation of these dams as prescribed by the 1995 Biological Opinion (NMFS 1995) and was therefore most similar to a status quo action. This action assumed that all of the measures called for in the 1995 Biological Opinion are implemented in full in every year. With action A2, the hydropower system is operated to collect as many fish as possible at the uppermost dams for transport by barge past the hydropower system and release below Bonneville Dam (the last dam encountered by juvenile salmon as they migrate to the ocean). Action A3 was drawdown to natural river levels of the four dams on the lower Snake River (Lower Granite, Little Goose, Lower Monumental, and Ice Harbor). Under this action, a 222-km unimpounded reach would be created by removing earthen berms adjacent to these dam structures. Hydroelectricity generation and navigation at these projects would be eliminated. We also evaluated three variations on actions A1, A2, and A3 but do not report their results here because

Fig. 1. Schematic diagram of the life cycle of Snake River spring and summer chinook salmon. The passage models estimated survival of smolts in the Columbia and Snake rivers as they pass through (in-river smolts) or around (transported smolts) the eight dams of the Federal Columbia River Power System (FCRPS). Transportation models estimated the relative estuary and ocean survival rate of transported and in-river smolts. Life-cycle models estimated survival rates over the rest of the life cycle (spawners in natal streams to smolts at the top of the hydropower system and from smolts at the bottom of the hydropower system through estuary and ocean residence and upstream migration back to spawners in their natal streams).



they produced similar outcomes to the three primary actions (Marmorek et al. 1998*c*).

Performance measures used to evaluate the actions

Performance measures provided a quantitative indicator of how well alternative management actions performed, both relative to other actions and with respect to absolute goals for survival and recovery of listed stocks. PATH used the "jeopardy standards" generally accepted by NMFS for Snake River chinook salmon (NMFS 1995). There were two types of standards: survival and recovery. The survival standard was based on the probability (over 3000 Monte Carlo simulations) that the projected number of spawners for individual stocks would exceed survival escapement thresholds of either 150 or 300 spawners. These probabilities were calculated over 24 and 100 simulation years to reflect short- and long-term effects. The recovery standard was based on the probability that the projected spawning abundance of a stock will be above a specified recovery escapement threshold (thresholds for the seven index stocks ranged from 350 to 1150 spawners). This probability was calculated over the last 8 years of a 48-year simulation. As an absolute standard for evaluating actions, PATH specified that most stocks should have survival probabilities of at least 0.7 (i.e., stocks exceed their survival escapement threshold in at least 70% of the simulated years) and recovery probabilities of at least 0.5. See the Appendix for details on how these probabilities were calculated.

Models to project outcomes of actions

The PATH modeling framework for the decision analysis used passage, transportation, and life-cycle models to represent the entire life cycle of spring and summer chinook salmon (Fig. 1). The modeling analysis proceeded in two linked steps: retrospective and prospective (Fig. 2). Retrospective modeling captured our understanding of the past by generating distributions of historical estimates of key model parameters. Prospective modeling used these distributions of historical estimates as the basis for assessing the range of possible future outcomes of management actions.

Retrospective passage and transportation modeling

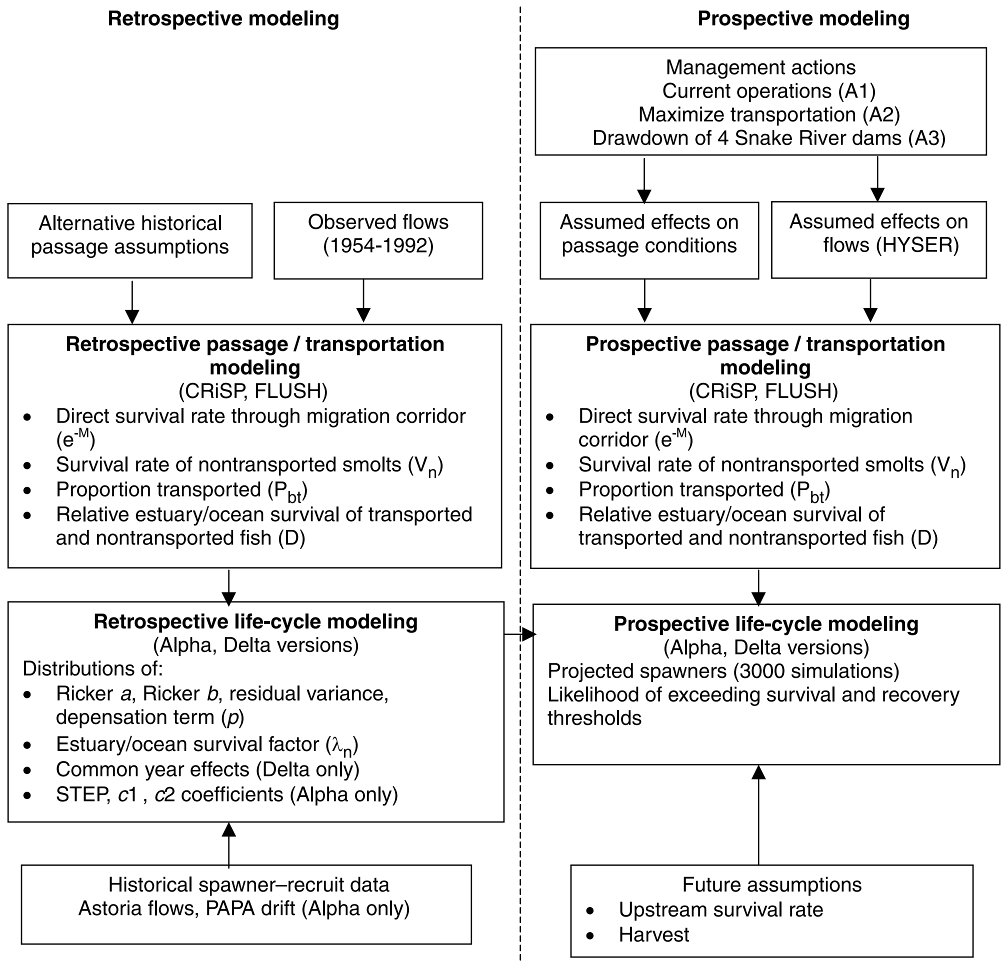
Retrospective passage and transportation modeling generated historical estimates (for brood years 1952–1990) of four parameters related to the passage of smolts through the migration corridor: (*i*) direct passage survival rate (e^{-M}, where M is the instantaneous mortality rate through the migration corridor), the weighted average survival rate of transported smolts and in-river smolts (smolts that are not collected for transportation but are allowed to migrate through the dams and reservoirs of the FCRPS), measured from the top of the reservoir of the first dam encountered (Lower Granite) to below the last dam (Bonneville); (*ii*) survival rate of in-river smolts from the head of Lower Granite reservoir to below Bonneville Dam (V_n); (*iii*) proportion of all smolts arriving below Bonneville Dam that were transported (P_{bt}); and (*iv*) estuary and ocean survival rate of transported fish relative to that of in-river fish (this ratio is called D; see Appendix for details on how D was estimated). $D < 1$ suggests that transported fish have lower estuary and ocean survival rates than in-river fish, whereas $D > 1$ suggests that transported fish survive better than in-river fish through this life stage.

Estimates of passage and transportation parameters depended on assumptions about historical passage conditions (such as the mortality of fish at specific dams in past years), the success of past transportation experiments, and observed flows over the historical period. The terms e^{-M}, P_{bt}, and D were combined to produce an estimate of system survival (ω), the weighted average survival rate of all smolts (transported and in-river) through the hydropower system with the survival rate of transported fish discounted by their hypothesized survival disadvantage in the estuary and ocean (see Appendix for details).

Retrospective life-cycle modeling

Estimates of the four passage and transportation parameters were

Fig. 2. Modeling approach used in the decision analysis. Retrospective modeling (left side of diagram) described and explained historical patterns in spawner-to-recruit survival rates. Results were expressed as distributions of parameters, which were passed to the prospective models. In the prospective phase (right side of diagram), retrospective results were used to forecast the response of stocks to alternative management actions under a range of assumptions and hypotheses. See text for definition of terms.



passed to the life-cycle model, along with historical spawner–recruit estimates (Beamesderfer et al. 1997) and environmental data (e.g., climate indices) from brood years 1952–1990. The life-cycle model was a generalized version of a Ricker stock–recruit model. We developed two versions of this model: an “Alpha” version (Anderson and Hinrichsen 1997) and a “Delta” version (Deriso et al. 2001). The two versions embodied different assumptions about the existence of common mortality effects that affect both Snake River and lower Columbia River stream-type chinook stocks and used different methods for calculating estuary and ocean survival factors for in-river fish (λ_n). The Delta model assumed that Snake and lower Columbia stocks experience common mortality effects and calculated estuary and ocean survival factors indirectly from estimated overall (spawner to recruit) survival rates. The Alpha model allowed for different responses to ocean climate and other factors in different regions within the Columbia River basin and calculated estuary and ocean survival rates directly from various climatic factors such as Columbia River flows at its mouth. Both versions estimated posterior probabilities of model parameters. The two versions are described further in the Appendix.

Prospective passage and transportation modeling

Prospective passage and transportation modeling projected future values for the four passage-related parameters (M, V_n, P_{bt}, D) based on assumptions about how the alternative management actions would affect flows and conditions that determine passage survival. Such conditions included the efficiency of bypass systems that route fish around turbines, the amount of spill at individual projects, and dam configurations and operations in future years. A hydro-regulation model (HYSER) developed by the U.S. Army Corps of Engineers translated each management action into mean monthly flows at various locations in the Snake and Columbia rivers, given the natural runoff conditions that were observed from 1977 to 1992. These years were chosen because they were most representative of current operations and conditions. The HYSER model generated 16 sets of mean monthly flows, one for each of these years. The passage models projected a set of the four passage-related parameters for each action, for each of the assumptions about the effect of that action on passage conditions, and for each year of monthly mean flows generated by the HYSER model for that action. For example, a parameter set projected with 1982 mean monthly flows represented the passage- and transportation-related survival rates expected to prevail if a given action were implemented, if that action had the assumed affect on passage conditions and flows, and if 1982 runoff conditions occurred.

Prospective life-cycle modeling

The life-cycle model combined posterior probability distributions of life-cycle parameters from the retrospective modeling, sets of downstream passage parameters from the prospective passage modeling, and assumptions about other future influences on survival (i.e., adult survival during upstream migration, harvest rates, habitat changes) into projections of spawning abundance for each of the seven Snake River index stocks over a 100-year simulation period beginning in brood year 1996 (Deriso 2001). In each of the 3000 Monte Carlo simulations, a set of life-cycle model parameters (i.e., Ricker *a*, Ricker *b*, *p*, residual variance) was selected from the distributions generated from the retrospective analyses. In each year of a simulation, the model also selected annual estuary and ocean survival factors (λ_n) and climate-related survival factors according to alternative hypotheses about how these factors would change in response to management actions (see next section), and a particular "flow year" from 1977 to 1992 according to how frequently the natural runoff in each of those years occurred in the 1929–1992 historical record. The selected flow year determined which of the 16 sets of prospective passage-related parameters was used for that year of the simulation.

Uncertainties that determine the range of possible outcomes for each action

Uncertainty in Ricker *a*, Ricker *b*, and other life-cycle parameters was incorporated into the forward projections through Monte Carlo analyses that sampled from the joint posterior distribution of these parameters. PATH scientists also included 11 additional discrete uncertainties in the decision analysis (Table 1). These uncertainties were selected either because they were major assumptions that PATH retrospective analyses had not sufficiently resolved because of data limitations or differences in interpretation of data or because they were related to future conditions and thus not addressed by the retrospective analyses. For each uncertainty, we formulated two or three alternative hypotheses about how the uncertainty was expected to affect the outcomes of actions. Alternative hypotheses took the form of different parameter values in the passage–transportation or life-cycle models, or different approaches to selecting from historical distributions of parameters in forward projections. Further details and rationale for all hypotheses can be found in Marmorek et al. (1998*a*, 1998*b*).

Juvenile passage survival rate

Values of e^{-M}, V_n, and P_{bt} were computed by two passage models: CRiSP (Columbia River Salmon Passage model, developed by researchers at the University of Washington for the Bonneville Power Administration; Anderson et al. 1996), and spring FLUSH (Fish Leaving Under Several Hypotheses, developed by State and Tribal researchers; Wilson 1994).

Fish guidance efficiency

Fish Guidance Efficiency (FGE) measures the ability of submerged screens to divert fish away from turbines into bypass systems with lower mortality rates than turbines. Extended-length screens have been installed at some dams in recent years, but their relative effectiveness in diverting fish to the bypass systems was uncertain.

Turbine and bypass mortality

Current estimates of turbine and bypass mortality rates for spring and summer chinook smolts are generally accepted, but there was uncertainty about the importance of direct physical injury (descaling) in causing mortality at some dams before 1980.

Predator-removal effectiveness

Since 1990, bounties have been offered for catching northern pikeminnow (*Ptychocheilus oregonensis*), a predator of migrating salmon smolts in reservoirs along the Columbia and Snake rivers. Uncertainty about the degree of compensatory responses of predator populations to the removal program led to disagreement about the effect of the program on survival rates of salmon smolts.

Transportation assumptions

Transportation assumptions were important because under current operating regimes, over 90% of smolts below Bonneville Dam arrive there by barge. Hypotheses about the effectiveness of transporting fish past the hydrosystem focused on the survival rate of transported fish, once they are released from the barge, relative to in-river fish (represented by *D* values). Each passage model developed its own set of historical and future *D* values (see details in Appendix) based on interpretations of results of past transportation and survival studies (Fig. 3). Future *D* values depended on assumptions about the applicability of historical transportation conditions to future simulations.

Regional mortality effects

There is uncertainty about the existence of common mortality effects that influence both Snake River and lower Columbia River stream-type chinook stocks (Schaller et al. 1999, 2000; Zabel and Williams 2000). The two versions of the life-cycle model (Alpha and Delta) embodied different assumptions about these effects and used different methods for calculating estuary and ocean survival factors for in-river fish (see Appendix).

Future climate

Short- to medium-term variation in future climate and its effects on survival rates was modeled by drawing from the historical distributions of climate factors estimated from retrospective modeling. Two sampling approaches were used to reflect possible patterns in future climate conditions. Longer-term climate cycles were considered in the Regime Shift hypothesis (described below).

Estuary and ocean survival factor for in-river fish (λ_n)

Historical distributions of estuary and ocean survival factors (λ_n) were estimated by the life-cycle model through retrospective modeling. These estimates were based on observed patterns in survival to adulthood of the seven Snake River index stocks (which migrate past 8 mainstem projects) and six conspecific downstream "control" stocks (which migrate through 1, 2, or 3 projects) (Deriso et al. 2001; Schaller et al. 1999). Between the pre-1970 and post-1974 periods, the survival rate of Snake River stocks declined significantly more than the downstream stocks. This suggested that some fundamental change in survival conditions occurred in the mid-1970s and that the effect of this change on Snake River fish was systematically different from the effect on downstream stocks.

The decline in survival rates in the mid-1970s coincided with several major changes in environmental and anthropomorphic influences on salmon, including the completion of the final two Federal dams on the lower Snake River, a substantial increase in hatchery production, and a shift in climate conditions in the Northeast Pacific Ocean. We developed three hypotheses (Hydro, Stock Viability, and Regime Shift) to account for possible effects of these changes on estuary and ocean survival rates of Snake River fish. These hypotheses provided alternative explanations for historical patterns in survival and had implications for how λ_n was expected to change in response to future management actions. For example, the Hydro explanation for historical survival patterns implied that the survival factor λ_n would increase significantly when dams were breached under the drawdown action (A3). More details on these hypotheses and how they were implemented in the life-cycle model are provided in the Appendix.

Duration of pre-removal period

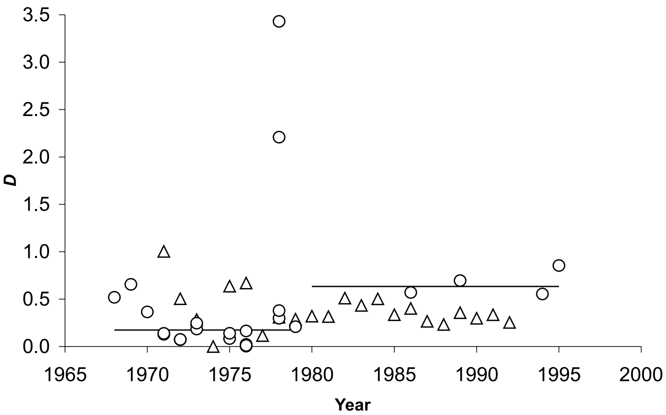
The pre-removal period represented the length of time between a decision to proceed with drawdown and the start of construction work to breach dams. Assumptions about the duration of this period depended on the expected length of the Congressional approval and appropriations process and the possibility of litigation.

Table 1. Uncertainties and alternative hypotheses incorporated into the PATH decision analysis.

Uncertainty	Alternative hypotheses
Total passage mortality (M), in-river survival rate (V_n), and proportion transported (P_{bt})	CRiSP passage model: mechanistic model with a daily time step. CRiSP survival rates through the FCRPS generally higher than FLUSH values. Spring FLUSH passage model: more aggregated, empirical approach with a seasonal time step.
Fish guidance efficiency (FGE)	Extended-length screens increase FGE. Extended-length screens do not increase FGE.
Turbine and bypass mortality before 1980	Higher turbine and bypass mortality: assumed turbine and bypass mortality entirely due to descaling. Lower turbine and bypass mortality: assumed turbine and bypass mortality partially due to descaling.
Predator-removal effectiveness	0% reduction in reservoir mortality: assumed high compensatory response by predators. 25% reduction in reservoir mortality: assumed low compensatory response by predators.
Transportation assumptions (D)	Transportation assumptions associated with FLUSH passage model: low survival rates of transported fish relative to in-river fish (median post-1980 D = 0.34) because of increased stress, deterioration of homing ability, and other mechanisms (Budy et al. 2002). Transportation assumptions associated with CRiSP passage model: high survival rates of transported fish relative to in-river fish (median post-1980 D = 0.63).
Regional mortality effects	Delta model: upstream and downstream stocks experience common climate effects Alpha model: upstream and downstream stocks respond independently to climate effects
Future climate Delta version: represented by δ (common year-effect) Alpha version: represented by E (latitude of a drifting object 3 months after release from ocean station PAPA in the Northeast Pacific Ocean)	Future simulations sample δ (Delta version) or E (Alpha version) either (*a*) from good or bad historical periods according to an 18.5-year cycle in ocean conditions (Beamish et al. 1999) or (*b*) with first-order autocorrelation from historical distributions for 1950–1995 (a period with both good and bad climatic conditions).
Estuary and ocean survival factor for in-river Snake River fish. Delta version: represented by λ_n Alpha version: represented by STEP	Hydro: estuary and ocean survival factor related to delayed effects of passage through the eight dams of the Federal Columbia River Power System (Budy et al. 2002). Stock viability: current (1977–1992) estuary and ocean survival rates will continue indefinitely in the future, even if hydrosystem direct mortality declines or climate improves. Regime shift: estuary and ocean survival factor follows long-term (60-year) cycle in ocean climate that affected Snake River fish more severely than downstream stocks after the mid-1970s.
Duration of pre-removal period under drawdown	3 years 8 years
Juvenile survival rate after drawdown	Survival rate through drawndown Snake River reach = 0.96 (average Snake River survival rate estimated before three of four Snake River dams were built; Raymond 1979); survival rates through the entire Snake and Columbia rivers (with only four dams in place) = 0.64 (CRiSP) and 0.56 (FLUSH). Assumed drawdown eliminates direct dam mortality, reduces reservoir habitats favored by predators, and restores more natural flows. Survival rate through drawndown Snake River reach = 0.85 (current estimate of average survival rates through the free-flowing stretch of the Snake River above Lower Granite Dam); survival rates through the entire Snake and Columbia rivers (with only four dams in place) = 0.55 (CRiSP) and 0.50 (FLUSH). Assumed drawdown only eliminates direct dam mortality; changes in predators and flow regimes since placement of dams are permanent and not affected by drawdown.
Duration of transition period after drawdown	2 years: processes equilibrating in a relatively short period of time (e.g., sediment removal, redistribution of predators) are more important for juvenile survival rates. 10 years: processes taking longer to equilibrate (stabilization and recolonization of banks, population responses by predators) are more important.

Note: Alternative hypotheses took the form of different parameter values in the passage–transportation or the life-cycle models or different approaches to selecting from historical distributions of parameters in forward projections. CRiSP, Columbia River Salmon Passage; FCRPS, Federal Columbia River Power System; FLUSH, Fish Leaving Under Several Hypotheses.

Fig. 3. Retrospective *D* values estimated by the FLUSH (Fish Leaving Under Several Hypotheses, triangles) and CRiSP (Columbia River Salmon Passage, circles) transportation models. See Appendix for details of each model's method. CRiSP values used in the retrospective analysis for periods 1968 to 1979 and 1980 to 1992 (horizontal lines) were the median of CRiSP estimates during those periods (0.174 and 0.633, respectively).



Juvenile survival rate after drawdown

Once implemented, drawdown of dams was assumed to eventually improve juvenile survival rates by eliminating direct dam mortality, reducing predation mortality in reservoirs, and restoring more natural flows. Because the extent to which all of these benefits would be realized was uncertain, we developed two hypotheses about post-drawdown juvenile survival rates. Under these hypotheses, post-drawdown survival rates of juveniles through the entire Snake and Columbia rivers (with four dams in place) ranged from 0.50 to 0.64 depending on the hypothesis and passage model. In comparison, average in-river survival rates through the existing hydropower system (i.e., all eight dams in place) were 0.24 (with FLUSH) and 0.39 (with CRiSP).

Duration of transition period

The transition period was the length of time between completion of dam removal and establishment of a dynamic equilibrium in the drawndown section of the river. Although there was general agreement that initially dam drawdown would likely disrupt physical and biological processes in the river, there were no case studies to suggest how long it would take for these processes to equilibrate.

Probabilities on alternative hypotheses

Decision analysis requires probabilities on the alternative hypotheses that reflect their relative likelihood, or strength of empirical support. Initially, for the purposes of conducting sensitivity analyses and seeing which of the uncertainties had the most influence on the outcomes of actions, we assumed that all hypotheses were weighted equally. However, we recognized that some hypotheses had more empirical support than others. Therefore, after completing preliminary sensitivity analyses and identifying key uncertainties (i.e., those most affecting modeled outcomes), we initiated a Weight of Evidence process to systematically compile the evidence and assess the relative merits of alternative hypotheses (appendix 4 in Marmorek and Peters 2001). Weights on alternative hypotheses were then elicited from four scientists on the independent PATH Scientific Review Panel based on the evidence available as of August 1998 (Table 2). Since 1998, new data and analyses have become available that may affect the weights assigned to alternative hypotheses. Therefore, it may be worthwhile to systematically compile and evaluate this new information and elicit a new set of weights from independent scientists. The PATH Weight of Evidence process provides an example of a collaborative approach to assessing evidence.

Decision analysis framework

We used a "decision tree" as a graphical tool for integrating actions, performance measures, models, and uncertainties into a logical evaluation framework (a highly simplified form of the PATH decision tree is shown in Fig. 4). The 11 uncertainties and their alternative hypotheses resulted in 240 possible outcomes for actions A1 (current operations) and A2 (maximize transportation) and 1920 possible outcomes for action A3 (natural river drawdown of four Snake River dams). Each outcome was based on 3000, 100-year prospective Monte Carlo simulations, each selecting a different set of life-cycle model parameters (Ricker *a*, Ricker *b*, *p*, residual variance) from their joint probability distribution.

These sets of outcomes provided the basis for identifying key uncertainties and evaluating the performance of actions across uncertainties. To identify which uncertainties had the most influence on outcomes, we applied a Categorical Regression Tree (CART; Breiman et al. 1984; Watters and Deriso 2000) analysis to the complete set of outcomes for all actions. The CART analysis used a sum-of-squares criterion to find the factors that explain most of the overall variation in the set of outcomes ($R^2 \geq 0.95$). We used the frequency distribution of outcomes (summarized with box-and-whisker plots) as an indication of how well each action performed over the full range of uncertainties.

Sensitivity analyses

Although PATH's primary focus was on evaluating hydrosystem actions, the PATH modeling structure was sufficiently flexible to allow consideration of non-hydro mortality factors such as freshwater spawning and rearing habitat, mainstem and tributary harvest, and avian predation in the estuary. Decision analysis provided a convenient framework for exploring the effects of hypothetical actions in these non-hydro areas. We do not report the results of these analyses here because we found that the effects of these non-hydro actions were generally not as significant as effects of hydrosystem actions, unless dramatic changes were made (e.g., permanent elimination of all mainstem commercial and sport fisheries and conservation-level tribal ceremonial fisheries) (Marmorek et al. 1998*c*).

Results

Relative effects of uncertainties

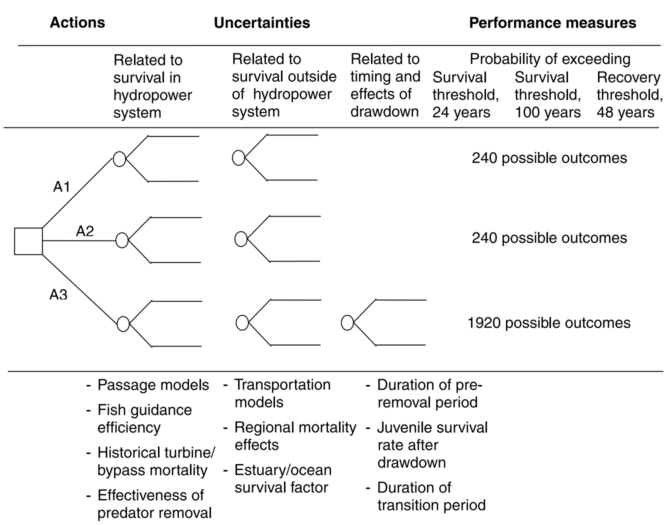
Assumptions associated with the alternative actions were most important in determining the 100-year probability of exceeding the survival threshold and the 48-year probability of exceeding the recovery threshold (Fig. 5; the CART tree for the probability of exceeding the survival escapement threshold over 100 years is similar to the CART tree for the 48-year recovery metric). That is, the effects of differences in the operation and configuration of the hydropower system associated with the different actions (e.g., differences between current and post-drawdown in-river survival rates) were more important in determining stock recovery than the effects of the 11 uncertainties described in Table 1. Of the uncertainties, regional mortality effects (Alpha vs. Delta models) and the cause of estuary and ocean mortality (hydro vs. stock viability vs. regime shift) were most important for

Table 2. Weights placed on alternative hypotheses by the four members of the PATH Scientific Review Panel (SRP).

Key uncertainty	Alternative hypothesis	Relative weights 1	2	3	4
Passage–transportation models	FLUSH	0.7	0.75	0.9	0.65
	CRiSP	0.3	0.25	0.1	0.35
Estuary and ocean survival factor for in-river fish	Stock viability	0.3	0.25	0.495	0.4
	Hydro	0.6	0.60	0.495	0.4
	Regime shift	0.1	0.15	0.010	0.2
Regional mortality effects	Alpha life-cycle model	0.0	0.7	0.1	0.1
	Delta life-cycle model	1.0	0.3	1.0	0.9
Length of transition period	2 years	0.6	0.33	0.2	0.5
	10 years	0.4	0.67	0.8	0.5
Turbine and bypass mortality before 1980	Upper bound	0.6	0.4	0.4	0.5
	Lower bound	0.4	0.6	0.6	0.5
Predator-removal effectiveness	0% reduction in reservoir mortality	0.7	1.0	0.8	0.9
	25% reduction in reservoir mortality	0.3	0.0	0.2	0.1
Juvenile survival rate after drawdown	0.85	0.6	0.8	0.8	0.25
	0.96	0.4	0.2	0.2	0.75

Note: Weights reflect each SRP member's judgement on the relative likelihood of each hypothesis, based on their assessment of the evidence provided by PATH (Marmorek et al. 1998*b*). Alternative hypotheses are described in the text and in Table 1. FLUSH, Fish Leaving Under Several Hypotheses; CRiSP, Columbia River Salmon Passage.

Fig. 4. Decision tree summarizing the decision analysis of recovery actions for Snake River spring and summer chinook salmon. The three actions to evaluate emanate from the "decision node" (square box) on the left. A1 is current operations, A2 is maximize transportation, and A3 is natural river drawdown of four lower Snake River dams. The circles represent "chance nodes", with branches indicating alternative hypotheses about the possible effects of each uncertainty on the outcomes. Each pathway along the branches represented a particular combination of alternative hypotheses, each of which produced a unique set of performance measures (probability of exceeding survival escapement thresholds over 24 and 100 years and probability of exceeding recovery escapement thresholds over 48 years) as shown on the right side of the tree. The tree is highly simplified for illustrative purposes (i.e., not all possible pathways are shown).



the drawdown action A3 (right side of Fig. 5), and passage–transportation models (CRiSP vs. FLUSH) were most important for actions A1 (current operations) and A2 (maximize transportation) (left side of Fig. 5). Further sensitivity analyses of the two passage and transportation models showed that the key difference between them was in their estimates of D (particularly the retrospective estimates), rather than in their estimates of direct hydropower survival (M, V_n, or P_{bt}; Marmorek et al. 1998*b*).

Uncertainties related to the cause of mortality of in-river fish in the estuary and ocean and the effectiveness of transportation (D) were the most important factors in determining the 24-year probability of exceeding the survival threshold (Fig. 6) and generally had more influence on outcomes than the actions. The projected escapement of stocks in the short term depended more on assumptions about these components of stock dynamics than on what actions were taken because drawdown actions were assumed to generate their maximum survival benefits only after 7–20 years (i.e., after the pre-removal, construction, and transition periods).

Performance of actions across uncertainties

Actions A1 (current operations) and A2 (maximize transportation) produced similar distributions of probabilities of exceeding survival and recovery thresholds (Fig. 7). A1 outcomes were actually slightly higher than A2 because spill (which generally produces a higher survival rate for smolts than going through turbines) was eliminated under A2 at several dams, whereas the proportion of fish transported in A2 was only slightly higher than that in A1. Means and medians of the distributions for these two actions were slightly below the standards for recovery (0.5) over 48 years and survival (0.7) over 24 years but slightly over the survival standard over 100 years. Around 65% of the 240 possible outcomes for both actions were below the survival standard of 0.7 over 24 years, 30% of the outcomes were below the 0.7 survival standard over 100 years, and 55% were below the 0.5 recovery standard over 48 years.

Outcomes for action A3 (natural river drawdown of four lower Snake River dams) were generally higher than for the other actions and had narrower distributions. Mean and median probabilities of exceeding the survival thresholds barely exceeded the 0.7 standard over 24 years but were well over the survival standard over 100 years and the 0.5 recovery standard over 48 years. Forty-five percent of the 1920 possible outcomes for A3 were below the 0.7 survival standard over 24 years, but virtually none of the outcomes was below the longer-term survival and the recovery standard.

Discussion

The PATH decision analysis helped to focus scientific debate on the key underlying hypotheses that account for differences in the results and implied recovery strategies of alternative modeling systems. Specifically, we found that the uncertainties that had the largest influence on the relative outcomes of actions were those related to the patterns and causes of mortality of Snake River chinook in the estuary and ocean life stage (e.g., relative estuary and ocean survival rates of transported and in-river fish *(D*), factors affecting estuary and ocean survival factors of in-river fish (λ_n), and regional patterns in estuary and ocean mortality effects). These uncertainties exist because rearing, migration, and estuary and ocean conditions changed at the same time that survival rates of Snake River fish began to decline dramatically.

Uncertainty about the cause of mortality in the estuary and ocean is unlikely to be resolved in the near future because existing monitoring programs provide equivocal information on survival rates through this life stage. For example, adult returns have been too low to provide reliable estuary and ocean survival estimates using PIT (passive induced transponder) tags. Similarly, recoveries of coded-wire tags have been too few to provide unambiguous information on ocean distributions of Snake River and downstream stream-type chinook stocks (Paulsen and Fisher 1997; Marmorek et al. 1998*b*). Such information is necessary to determine the importance of ocean conditions in observed differences in historical patterns of ocean survival rates between different stock groups (Schaller et al. 1999). Transportation studies using PIT-tagged groups of transported and in-river groups of fish are designed to yield specific information on relative smolt-to-adult survival rates of these different groups and have been conducted since the late 1960s. However, the time series of data from these studies is confounded by changes in study methods over time (e.g., freeze branding vs. PIT tagging, trucking vs. barging). Current transportation studies are much improved over previous methods but still require various assumptions and produce distributions of *D* estimates under current, but not historical, conditions (NMFS 1999; Bouwes et al. 1999). Sensitivity analyses showed that historical *D* estimates were an important determinant of model outcomes.

Although new data and analyses are not likely to resolve key uncertainties, they may lead to the formation of new alternative hypotheses and weighting schemes. The PATH decision analysis provides a convenient framework for exploring the implications of additional hypotheses and alternative weighting schemes. However, these new hypotheses and weights should be developed and evaluated based on all available empirical evidence, not on their effects on projected outcomes of alternative recovery strategies.

Constraints on current monitoring programs, coupled with the confounding associated with implementing many different management actions at the same time, suggest that the best and perhaps only way to resolve key uncertainties is to implement large-scale management experiments and concurrent monitoring of survival rates over both the whole life cycle and specific life stages. PATH has completed a preliminary evaluation of some potential management experiments, the results of which indicate that well-designed experiments can reduce confounding and provide information on the factors that affect both overall and life stage specific survival rates (Peters et al. 2000; C. Paulsen, 16016 SW Boones Ferry Road, Lake Oswego, OR 97035, unpublished data). Although it is often difficult to implement experimental management with endangered species because of regulatory and logistical restrictions, the need for experimental management can be greater in these situations because the high political and social costs of decisions regarding endangered species generally cause decision makers to demand higher levels of certainty than are achievable with standard monitoring approaches.

Given the existence of several key uncertainties unlikely to be fully resolved in the short term, a biologically risk-averse approach to choosing a recovery strategy is to implement a “robust” action that achieves survival and recovery goals over a broad range of assumptions. Such an approach avoids relying on an unlikely or infrequent set of conditions to recover the stocks and thus minimizes the likelihood of “surprise” outcomes over the long term. Moreover, explicitly including uncertainties in the analysis of alternative actions allows the identification of a robust course of action now, without having to wait for these uncertainties to be resolved. Our results suggest that natural river drawdown of the four lower Snake River dams was the most robust action, meeting the 100-year survival and 48-year recovery goals over virtually all of the hypotheses and assumptions included in the decision analysis. This result suggests that we can be relatively confident that the drawdown action will meet long-term survival and recovery goals, even if we are not sure which of our hypotheses is correct. Neither drawdown nor the transportation action was able to achieve short-term survival targets with any certainty because of the low numbers of spawners in recent years.

As in any modeling exercise, we have included various assumptions, and excluded others, to represent a complex world in a simplified quantitative model. Although we have captured what we think are the most important components for evaluating hydrosystem actions, we acknowledge that the absolute predictive power of our models is probably low. Subsequent to the completion of the PATH modeling exercise, spawner estimates were generated for brood years 1996–1999. Predicted spawner numbers from our models for that period were generally higher than the actual estimates, suggesting that our modeled escapements were probably optimistic. The primary reason for this was that our simulations assumed that future climate conditions would resemble those from brood years 1952–1991 and conditions that determine passage-related mortality would resemble those from brood years 1975–1990. Both periods included good and bad conditions for salmon production, but the overall average conditions over the entire time periods were consider-

Fig. 5. Categorical Analytical Regression Tree (CART) diagram showing the relative influence of actions and uncertainties on the probability of exceeding the recovery escapement threshold over 48 years. Each branch on the tree represents a split between two or more factors (actions or alternative hypotheses) that account for differences in the outcomes. The left side of each split represents factors that result in lower probabilities of exceeding the survival or recovery threshold; the right side represents factors that produce higher probabilities. The most important factors (those that account for the greatest amount of differences in the results) are split first (at the top), followed by progressively less important factors at the bottom of the diagram. The vertical length of each branch is proportional to the amount of variance explained by each factor, and the number at the bottom of each branch is the average outcome for that combination of factors. The most important factors were assumptions relating to the actions themselves, the passage–transportation models, estuary and ocean survival factors of in-river fish, and regional patterns in estuary and ocean mortality effects.

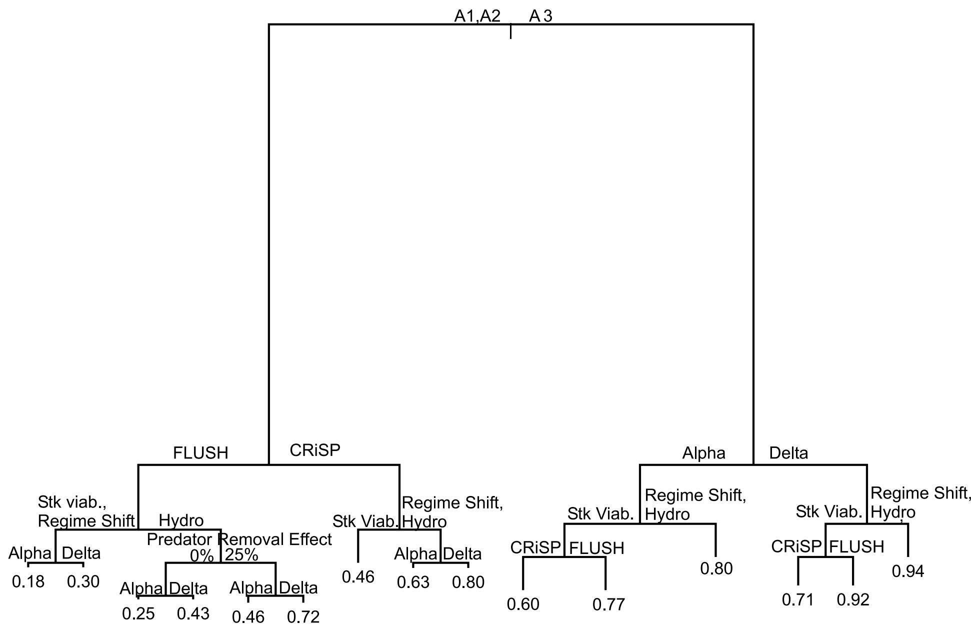
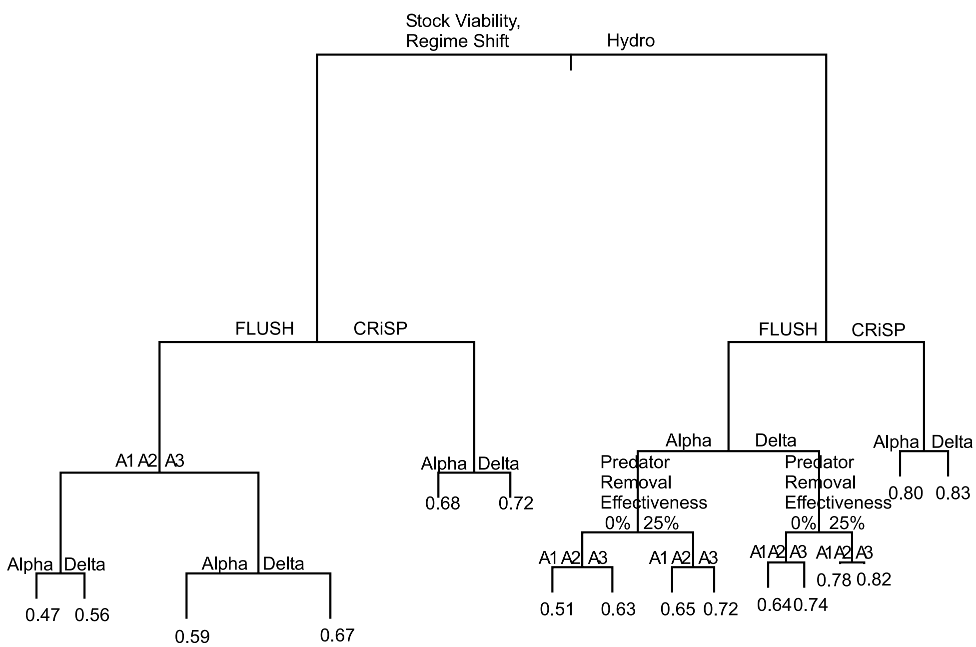
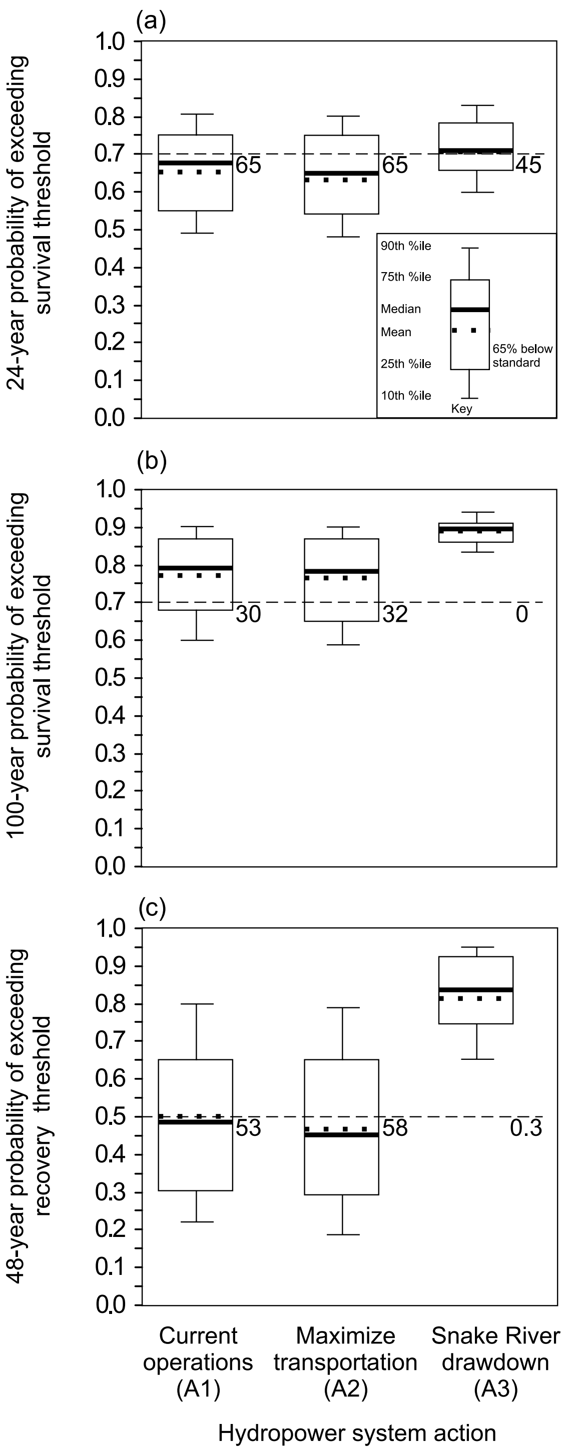


Fig. 6. Categorical Analytical Regression Tree (CART) diagram showing the relative influence of actions and uncertainties on the probability of exceeding the survival threshold over 24 years. See Fig. 5 for interpretation. The most important factors were the passage–transportation models, estuary and ocean survival factors for in-river fish, and regional patterns in estuary and ocean mortality effects. This CART diagram is truncated for clarity of presentation; if pruned to an R^2 of 0.95, there were 37 splits.



ably better for fish than conditions have been since 1995. The base periods that we used thus represented more favorable future climate and mortality conditions, and thus produced higher projected numbers of spawners, than what would be expected using only the most recent estimates of common year-effects and total mortality factors. Although

Fig. 7. Distributions of probabilities of exceeding (*a*) the survival threshold over 24 years, (*b*) the survival threshold over 100 years, and (*c*) the recovery threshold over 48 years for the three hydrosystem actions. The figure also shows the mean and median of the distributions and the percentage of outcomes that fell below the standards of 0.7 for survival thresholds and 0.5 for recovery thresholds (these standards are indicated by horizontal broken lines). There were 240 outcomes for actions A1 and A2 and 1920 for A3. Distributions of the 24-year survival metric were similar for the three actions. Over the long term (48 and 100 years), however, A3 tended to produce higher and less variable probabilities of exceeding survival and recovery thresholds than did A1 and A2.



this leads our models to be optimistic, it applies to all actions equally and thus will not affect the relative ranking of actions.

In conclusion, decision analysis provides a useful approach for including a broad range of alternative hypotheses about past and future responses of Snake River spring and summer chinook stocks to environmental and anthropogenic factors, systematically looking at the range of possible effects of recovery actions on these stocks, and assessing how robust each action is to uncertainties. By including these uncertainties, robust recovery strategies can be identified before uncertainties are fully resolved, which may take more time than listed stocks have. The PATH decision analysis showed that uncertainties related to the cause of mortality in the estuary and ocean life stages had the most influence on projected outcomes of actions to recover Snake River spring and summer chinook stocks. These uncertainties are more likely to be resolved by implementing well-designed management experiments than by status quo actions because of limitations in current monitoring approaches for measuring changes in estuary and ocean survival rates. Given that these uncertainties exist, the natural river drawdown option was more risk averse than either continuing current operations or maximizing transportation.

Acknowledgements

We thank all PATH participants who provided data and analyses in support of this work. Jim Anderson, Josh Hayes, Howard Schaller, Earl Weber, and Paul Wilson provided passage model runs. Rick Deriso developed the life-cycle model and conducted the CART analyses. The PATH Scientific Review Panel provided many comments and suggestions for improving the analyses. Ian Parnell, Charlie Paulsen, and two anonymous reviewers provided helpful comments on earlier drafts. This work was funded by Bonneville Power Administration through the Northwest Power Planning Council's Fish and Wildlife Program, project numbers 9600600, 9600800, 9600801, 9601700, 9800100, 9303701, 8910800, 9700200, 9800600, and 8910700.

References

Anderson, J.J., and Hinrichsen, R. 1997. Prospective analysis for the alpha model. *In* Plan for Analyzing and Testing Hypotheses (PATH): Retrospective and Prospective Analyses of spring/summer chinook reviewed in FY 1997. *Edited by* D.R. Marmorek and C.N. Peters, ESSA Technologies Ltd., Vancouver, B.C. *Available from* Bonneville Power Administration, Portland, Oregon (http://www.efw.bpa.gov/Environment/PATH/reports/1997retro/toc.htm).

Anderson, J., Hayes, J., and Zabel, R. 1996. Columbia River Salmon Passage model: theory, calibration and validation. Center for Quantitative Science, University of Washington, Seattle, Wash. (http://www.cqs.washington.edu/crisp/crisp.html).

Barnthouse, L.B. 1993. Expert initial review of Columbia River Basin salmonid management models: summary report. *Prepared by and available from* Oak Ridge National Laboratory, Oak Ridge, Tenn.

Barnthouse, L.B., Marmorek, D.R., and Peters, C.N. 2000. Assessment of multiple stresses at regional scales. *In* Multiple stressors in ecological risk and impact assessment: approaches to risk estimation. *Edited by* S. Ferenc and J. Foran. Society of Environmental Toxicology and Chemistry Press, Pensacola, Fla.

Beamesderfer, R.C.P., Schaller, H.A., Zimmerman, M.P., Petrosky, C.E., Langness, O.P., and LaVoy, L. 1997. Spawner–recruit data for spring and summer chinook salmon populations in Idaho, Oregon, and Washington. *In* Plan for Analyzing and Testing Hypotheses (PATH): Retrospective and Prospective Analyses of spring/summer chinook reviewed in FY 1997. *Edited by* D.R. Marmorek and C.N. Peters, ESSA Technologies Ltd., Vancouver, B.C. *Available from* Bonneville Power Administration, Portland, Oregon (http://www.efw.bpa.gov/Environment/PATH/reports/1997retro/toc.htm).

Beamish, R.J., Noakes, D.J., McFarlane, G.A., Klyashtorin, L., Ivanov, V.V., and Kurasho, V. 1999. The regime shift concept and natural trends in the production of Pacific salmon. Can. J. Fish. Aquat. Sci. **56**: 516–526.

Botsford, L.W., and Paulsen, C.M. 2000. Assessing covariability among populations in the presence of intraseries correlation: Columbia River spring–summer chinook salmon (*Oncorhynchus tshawytscha*) stocks. Can. J. Fish. Aquat. Sci. **57**: 616–627.

Bouwes, N., Schaller, H.A., Budy, P., Petrosky, C.E., Kiefer, R., Wilson, P., Langness, O.P., Weber, E., and Tinus, E. 1999. An analysis of differential delayed mortality experienced by stream-type chinook salmon of the Snake River. Tech. Rep. Oregon Department of Fish and Wildlife, Portland, Oregon.

Breiman, L., Friedman, J.H., Olshen, R.A., and Stone, C.J. 1984. Classification and regression trees. Chapman & Hall, New York.

Budy, P., Thiede, G.P., Bouwes, N., Petrosky, C.E., and Schaller, H. 2002. Evidence linking delayed mortality of Snake River salmon to their earlier hydrosystem experience. North Am. J. Fish. Manag. In press.

Clemen, R.T. 1996. Making hard decisions: an introduction to decision analysis. 2nd ed. Duxbury Press, Wadsworth Publishing Co., Belmont, Calif.

Deriso, R.B. 2001. Bayesian analysis of stock survival and recovery of spring and summer chinook of the Snake River basin. *In* Incorporating uncertainty into fishery models. *Edited by* J.M. Berkson, L.L. Kline, and D.J. Orth. American Fisheries Society, Bethesda, Md. In press.

Deriso, R.B., Marmorek, D.R., and Parnell, I.J. 2001. Retrospective patterns of differential mortality and common year-effects experienced by spring chinook salmon (*Oncorhynchus tshawytscha*) of the Columbia River. Can. J. Fish. Aquat. Sci. **58**: 2419–2430.

Hare, S.R., Mantua, N.J., and Francis, R.C. 1999. Inverse production regimes: Alaska and West Coast Pacific salmon. Fisheries, **24**: 6–14.

Marmorek, D., and Peters, C. 2001. Finding a path towards scientific collaboration: insights from the Columbia River Basin. Conservation Ecol. In press.

Marmorek, D.R., Anderson, J., Basham, L., Bouillon, D., Cooney, T., Deriso, R., Dygert, P., Garrett, L., Giorgi, A., Langness, O.P., Lee, D., McConnaha, C., Parnell, I., Paulsen, C.M., Peters, C.N., Petrosky, C., Pinney, C., Schaller, H.A., Toole, C., Weber, E., Wilson, P., and Zabel, R.W. 1996*a*. Plan for Analyzing and Testing Hypotheses (PATH): final report on Retrospective Analyses for fiscal year 1996. *Compiled and Edited by* ESSA Technologies Ltd., Vancouver, B.C. *Available from* Bonneville Power Administration, Portland, Oregon (http://www.efw.bpa.gov/Environment/PATH/reports/ISRP1999CD/PATH%20Reports/FY96_Retro_Report/).

Marmorek, D.R., Peters, C.N., Anderson, J., Barnthouse, L., Beamesderfer, R., Botsford, L., Cooney, T., Coutant, C., Deriso, R., Dygert, P., Geiselman, J., Giorgi, A., Jones, M., Langness, O., McConnaha, C., Paulsen, C.M., Peterman, R., Petersen, J., Petrosky, C., Pinney, C., Schaller, H., Smith, S., Toole, C., Weber, E., Williams, J., and Wilson, P. 1996*b*. PATH—Plan for Analyzing and Testing Hypotheses. Conclusions of FY 96 Retrospective Analyses. *Prepared by* ESSA Technologies Ltd., Vancouver, B.C. *Available from* Bonneville Power Administration, Portland, Oregon (http://www.efw.bpa.gov/Environment/PATH/cncdft10.html).

Marmorek, D.R., Peters, C.N., Anderson, J., Barnthouse, L., Botsford, L., Cooney, T., Deriso, R., Geiselman, J., Giorgi, A., Hayes, J., Hinrichsen, R., Jones, M., Krasnow, L., Langness, O., Lee, D., McConnaha, C., Paulsen, C.M., Peterman, R., Petrosky, C.E., Pinney, C., Promislow, M., Schaller, H., Smith, S., Toole, C., Weber, E., Williams, J., Wilson, P., and Zabel, R. 1998*a*. Plan for Analyzing and Testing Hypotheses (PATH): preliminary decision analysis report on Snake River spring/summer chinook. Draft Report, March 1998. *Prepared by* ESSA Technologies Ltd., Vancouver, B.C. *Available from* Bonneville Power Administration, Portland, Oregon (http://www.efw.bpa.gov/Environment/PATH/reports/pdar/index.html).

Marmorek, D.R., Peters, C.N., Anderson, J., Barnthouse, L., Botsford, L., Bouwes, N., Budy, P., Cooney, T., Deriso, R., Geiselman, J., Giorgi, A., Hayes, J., Hinrichsen, R., Jones, M., Krasnow, L., Langness, O., Lee, D., McConnaha, C., Paulsen, C.M., Peterman, R., Petrosky, C.E., Pinney, C., Schaller, H., Smith, S., Toole, C., Weber, E., Williams, J., Wilson, P., and Zabel, R. 1998*b*. Plan for Analyzing and Testing Hypotheses (PATH): Weight of Evidence report. *Compiled and edited by* ESSA Technologies Ltd., Vancouver, B.C. *Available from* Bonneville Power Administration, Portland, Oregon (http://www.efw.bpa.gov/Environment/PATH/reports/ISRP1999CD/ PATH%20Reports/WOE_Report/).

Marmorek, D.R., Peters, C.N., Parnell, I., Anderson, J., Botsford, L., Bouwes, N., Budy, P., Connor, B., Cooney, T., Deriso, R., Geiselman, J., Giorgi, A., Hayes, J., Hinrichsen, R., Jones, M., Krasnow, L., Langness, O., Lee, D., McConnaha, C., Muir, B., Norris, J., Paulsen, C.M., Peterman, R., Petersen, J., Petrosky, C., Pinney, C., Schaller, H., Smith, S., Thompson, B., Tinus, E., Toole, C., Weber, E., Williams, J., Wilson, P., and Zabel, R. 1998*c*. PATH final report for fiscal year 1998. *Prepared by* ESSA Technologies Ltd., Vancouver, B.C. *Available from* Bonneville Power Administration, Portland, Oregon (http://www.efw.bpa.gov/Environment/PATH/reports/1998Final/1998Final.pdf).

National Marine Fisheries Service (NMFS). 1995. Reinitiation of consultation on 1994–1998 operation of the Federal Columbia River Power System and Juvenile Transportation in 1995 and Future years. Northwest Region, National Marine Fisheries Service, Seattle, Wash.

National Marine Fisheries Service (NMFS). 1999. Appendix A (anadromous fish). *In* Improving salmon passage: DRAFT Lower Snake River Juvenile Salmon Migration Feasibility Report/Environmental Impact Statement. U.S. Army Corps of Engineers, Walla Walla District, Walla Walla, Wash. (http://www.nww.usace.army.mil/html/offices/pl/er/studies/lsrpublic/lsrmain.htm).

Paulsen, C.M., and Fisher,T. 1997. Update on ocean distribution of coded wire tagged spring/summer chinook. *In* Plan for Analyzing and Testing Hypotheses (PATH): Retrospective and Prospective Analyses of spring/summer chinook reviewed in FY 1997. *Edited by* D.R. Marmorek and C.N. Peters, ESSA Technologies Ltd., Vancouver, B.C. *Available from* Bonneville Power Administration, Portland, Oregon (http://www.efw.bpa.gov/Environment/PATH/reports/1997retro/toc.htm).

Peterman, R., and Anderson, J. 1999. Decision analysis: a method for taking uncertainties into account in risk-based decision making. Human Ecol. Risk Assess. **5**: 231–244.

Peters, C.N., Marmorek, D.R., Anderson, J., Bouwes, N., Budy, P., Cooney, T., Deriso, R., Giorgi, A., Hinrichsen, R., Muir, B., Norris, J., Parnell, I., Paulsen, C.M., Petrosky, C., Pinney, C., Smith, S., Weber, E., and Zabel, R. 2000. PATH preliminary evaluation of the learning opportunities and biological consequences of monitoring and experimental management actions. *Prepared by* ESSA Technologies Ltd., Vancouver, B.C. Bonneville Power Administration, Portland, Oregon (http://www.efw.bpa.gov/Environment/PATH/reports/2000_M_ and_E.pdf).

Peters, C.N., Marmorek, D.R., and Deriso, R.B. 2001. Application of decision analysis to evaluate recovery actions for threatened Snake River fall chinook salmon (*Oncorhynchus tshawytscha*). Can. J. Fish. Aquat. Sci. **58**: 2447–2458.

Petrosky, C.E., Schaller, H.A., and Budy, P. 2001 Productivity and survival rate trends in the freshwater spawning and rearing stage of Snake River chinook salmon (*Oncorhynchus tshawytscha*). Can. J. Fish. Aquat. Sci. **58**: 1196–1207.

Raymond, H.L. 1979. Effects of dams and impoundments on migrations of juvenile chinook salmon and steelhead from the Snake River, 1966 to 1975. Trans. Am. Fish. Soc. **108**: 505–529.

Schaller, H.A., Petrosky, C.E., and Langness, O.P. 1999. Contrasting patterns of productivity and survival rates for stream-type chinook salmon (*Oncorhynchus tshawytscha*) populations of the Snake and Columbia rivers. Can. J. Fish. Aquat. Sci. **56**: 1031–1045.

Schaller, H.A., Petrosky, C.E., and Langness, O.P. 2000. Reply to Zabel and Williams' comments on "Contrasting patterns of productivity and survival rates for stream-type chinook salmon (*Oncorhynchus tshawytscha*) populations of the Snake and Columbia Rivers" by Schaller et al. (1999). Can. J. Fish. Aquat. Sci. **57**: 1741–1746.

Ward, D.L., Boyce, R.R., Young, F., and Olney, F. 1997. A review and assessment of transportation studies for juvenile chinook salmon in the Snake River. North Am. J. Fish. Manag. **17**: 652–662.

Watters, G., and Deriso, R. 2000. Catch per unit effort of bigeye tuna: a new analysis with regression trees and simulated annealing. Inter-Am. Trop. Tuna Comm. Bull. **21**: 531–571.

Wilson, P. 1994. Spring FLUSH (Fish Leaving Under Several Hypotheses). Version 4.5. Draft documentation. July 15, 1994. Columbia Basin Fish and Wildlife Authority, Portland, Oregon.

Zabel, R.W., and Williams, J.G. 2000. Comments on "Contrasting patterns of productivity and survival rates for stream-type chinook salmon (*Oncorhynchus tshawytscha*) populations of the Snake and Columbia Rivers" by Schaller et al. (1999). Can. J. Fish. Aquat. Sci. **57**: 1739–1741.

Appendix. Technical supplement

Calculation of survival and recovery standards

The survival standard was based on the probability that the projected number of spawners of individual stocks will exceed escapement thresholds that are thought to be high enough to avoid extinction. For the seven Snake River spring and summer chinook index stocks, the threshold level was either 150 or 300 spawners, depending on the amount of spawning habitat in spawning streams (the higher threshold was applied to streams where spawners are spread out over a larger area). These values were chosen because below these escapement levels, spawner–recruit relationships are poorly known and unpredictable changes in population behavior (e.g., depensation) are likely to occur. The probability of exceeding the survival threshold for each action was calculated as the fraction of the total number of years simulated (3000 Monte Carlo simulations × number of years in each simulation) in which the projected escapement exceeded the threshold. Two time periods were used: simulation years 1 to 24, to reflect short-term implications of actions, and a 100-year simulation period to reflect long-term effects.

The recovery standard described the ability of a certain hydrosystem action to recover stocks to healthier levels. This standard was based on the probability that the spawning abundance of a stock will be above a specified recovery escapement threshold. For spring and summer chinook stocks, the recovery escapement threshold was calculated as 60% of the pre-1971 brood-year average spawner counts for each stock (thresholds for the seven index stocks ranged from 350 to 1150 spawners). The probability of exceeding the recovery threshold was calculated by first taking the geometric mean spawner abundance over the last 8 years of a 48-year simulation (to dampen the effects of unusual years) and then calculating the fraction of the 3000 Monte Carlo simulations in which that geometric mean exceeded the recovery escapement threshold.

The probabilities of exceeding survival and recovery thresholds were calculated for each of the seven Snake River index stocks of spring and summer chinook. To aggregate these performance measures for individual stocks into a single measure for the entire Snake River spring and summer chinook ESU, NMFS defined an overall jeopardy standard. To meet this standard, an action must result in a "high percentage" of the individual populations having a "high" probability of being above the survival threshold level and a "moderate" probability of being above the recovery level. NMFS defined "high percentage" of stocks as 80% of the index populations. "High" and "moderate" probabilities were informally defined by PATH as 0.7 for the survival threshold and 0.5 for the recovery threshold. Therefore, for an action to meet the overall jeopardy standard, it must result in six out of the seven Snake River index stocks (around 80%) having at least a 0.7 probability of being above the survival threshold and at least a 0.5 probability of being above the recovery threshold. For this reason, we report probabilities of exceeding survival and recovery thresholds only for the sixth best stock.

Transportation assumptions

General approach for estimating D

D is the ratio of the estuary and ocean survival rate of transported fish (λ_t) to estuary and ocean survival rate of in-river (nontransported) fish (λ_n). D was estimated from smolt-to-adult survival rates (SARs) of experimental groups of tagged fish in transportation studies (Ward et al. 1997) using eq. A1:

$$\text{(A1)} \quad D = \frac{\lambda_t}{\lambda_n} = \frac{\mathrm{SAR}_t}{\mathrm{SAR}_n} \cdot \frac{V_c}{V_t} = T{:}C \cdot \frac{V_c}{V_t}$$

where λ_t is the estuary and ocean survival rate of transported fish, λ_n is the estuary and ocean survival rate of in-river fish, SAR_t is the smolt-to-adult survival rate of experimental groups of transported fish from the point of their collection and placement in a truck or barge back to that same point as adults, and SAR_n is the smolt-to-adult survival rate of experimental groups of in-river fish from the point of their collection and re-release back to that same point as adults. The ratio of SAR_t:SAR_n is known as the transport–control ratio (T:C). V_t is the survival rate of experimental groups of transported fish in the barge from the top of the hydropower system to below Bonneville Dam (assumed to be 0.98), and V_c is the survival rate of experimental groups of in-river fish from the top of the hydropower system to below Bonneville Dam (obtained from passage models or survival studies).

Transportation hypotheses

The decision analysis incorporated two hypotheses about the relative effect of transportation on survival rates in the estuary and ocean. Each hypothesis was associated with one of the two passage models. The first D hypothesis was implemented in conjunction with the FLUSH passage model. Historical D estimates were derived from a function relating T:C to the survival of in-river fish from the tailrace of the collection dam to below Bonneville dam (V_c). T:C data for this function came from transport studies conducted at Lower Granite and Little Goose dams in 1971, 1972, 1973, 1975, 1976, 1978, 1979, 1986, and 1989; estimates of V_c were based on per-project expansions of NMFS reach survival studies for these years except 1986 and 1989, when there were no NMFS survival studies (V_c was estimated with the FLUSH passage model in those years). The function was applied to retrospective FLUSH model estimates of V_c from 1977 to 1992 to derive estimates of T:C, from which annual D values were calculated using eq. A1. These annual values were used in the retrospective modeling. The median V_c value using this approach was 0.26 from 1968–1979 and 0.22 from 1980–1992. Median D values were 0.41 before 1980 and 0.34 from 1980–1992. The T:C vs. V_c relationship was also applied prospectively, in conjunction with projected estimates of V_c values under the various management actions, to derive a set of year-specific D values for prospective modeling.

The second D hypothesis was implemented with the CRiSP passage model. In this hypothesis, historical D values were estimated for each transportation study carried out at Lower Granite, Little Goose, Ice Harbor, and McNary dams between 1969 and 1995 using the T:C ratio estimated from that study, an estimate of the survival rate of in-river fish (V_c) from the CRiSP passage model, and eq. A1. Median V_c estimates using the CRiSP model were 0.13 before 1980 and 0.34 from 1980 to 1996. The median of the 1969–1979 D values (0.17) was used for retrospective modeling of years 1977–1979 and the median of the 1980–1995 D values (0.63) was used for retrospective modeling of years 1980–1992. For prospective modeling, the life-cycle model selected randomly from D values calculated from the most recent studies (1986, 1989, 1994, and 1995) because they were thought to be most representative of current collection and transport methods.

Mathematical description of life-cycle models

Delta version

The mathematical representation of the Delta version of the life-cycle model was

$$\text{(A2)} \quad \ln(R_{t,i}) = (1+p)\ln(S_{t,i}) + a_i - b_i S_{t,i} - M_{t,i} - \Delta m_t + \delta_t + \varepsilon_{t,i}$$

where $R_{t,i}$ (adult returns to the Columbia River mouth originating from spawning in year t and river sub-basin i) and $S_{t,i}$ (spawners in year t and river sub-basin i) were provided from the run reconstructions (Beamesderfer et al. 1997). $M_{t,i}$ was direct instantaneous passage mortality estimated by the passage models from the head of Lower Granite reservoir to below Bonneville Dam, which depended on year and included combined mortality of both transported and in-river smolts. $M_{t,i}$ varies for downriver stocks but is the same for all sub-basins i within the Snake River sub-region. Estimated parameters in eq. A2 were a_i (Ricker a parameter, inherent stock productivity for sub-basin i), b_i (Ricker b parameter, stock-carrying capacity for sub-basin i), and a depensation parameter (p), which represents a reduction in the number of recruits per spawner as spawner abundance declines and is applied to all stocks. Fits of eq. A2 to the spawner–recruit data generated values of p close to zero, which suggests that there is little evidence for depensation in these data. We also estimated δ_t (common year-effects influencing both Snake River and six Lower Columbia River stocks), $\varepsilon_{t,i}$ (random effects specific to each stock in each year), and Δm_t (extra mortality rate, which depends on year t). "Extra mortality" was any mortality occurring outside the juvenile migration corridor that was not accounted for by the other terms in eq. A2.

The Δm_t term was estimated as

$$\text{(A3)} \quad \Delta m_t = m_{t,i} - M_{t,i}$$

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

$$\text{(A4)} \quad m_{t,i} = X \cdot n_{t,i} + \mu_t$$

where $n_{t,i}$ is the total number of X-type dams (defined as Bonneville, John Day, and (or) The Dalles) that stock i must pass in year t, X is the dam passage mortality for all X-type dams and years (estimated by the life-cycle model), and μ_t is the incremental total mortality between the Snake River basin and the furthest up-river X dam in year t (also estimated by the life-cycle model).

The estuary and ocean survival rate of in-river fish (λ_n) was estimated retrospectively as

$$\text{(A5)} \quad \lambda_n = e^{(-m-\ln(\omega))}$$

where m was defined as in eq. A4 and ω (system survival) was defined as the number of in-river equivalent smolts below Bonneville Dam (i.e., adjusting for relative estuary and ocean survival of transported fish) divided by the population at the head of the first reservoir and was calculated as

$$\text{(A6)}\quad \omega = e^{(-M)}(DP_{bt} + 1 - P_{bt})$$

where e^{-M} is the weighted average survival rate of transported and in-river smolts from the top of the Lower Granite reservoir to below Bonneville dam (estimated by the passage models), P_{bt} is the proportion of all smolts arriving below Bonneville Dam that were transported (estimated by the passage models), and D is the estuary and ocean survival rate of transported fish relative to that of in-river fish (estimated by the transportation models; see eq. A1). The Delta version of the life-cycle model thus calculated λ_n as the residual left after factoring the system survival rate (ω) and a common year-effect (δ) from the estimated overall survival rate.

Alpha version

The basic equation for the Alpha version of the life-cycle model was

$$\text{(A7)}\quad \ln(R_{t,i}) = (1+p)\ln(S_{t,i}) + a_i - b_i S_{t,i} - M_{t,i} - \alpha_{t,j} + \varepsilon_{t,i}$$

where all terms in the Alpha model except $\alpha_{t,j}$ are the same as terms in the Delta model. The $\alpha_{t,j}$ term is the extra mortality in year t for subregion j:

$$\text{(A8)}\quad \alpha_{t,j} = \alpha_n - (\text{average } \alpha_n) - \ln(D_t P_{bt} + 1 - P_{bt}) + (\text{average } \ln(D_t P_{bt} + 1 - P_{bt}))$$

where the averaged terms encompass brood years 1952–1990, $P_{b,t}$ and D_t are defined as in eq. A6, and

$$\text{(A9)}\quad \alpha_n = (c_1 / F_t) + (c_2 E_t / F_t) + \text{STEP}_j$$

where F_t is the average flow (in 1000 ft^3/s) at the mouth of the Columbia River for year t during April and June and E_t is the PAPA drift climate index variable (the latitude of a drifting object after 3 months starting at Ocean Station PAPA in the Northeast Pacific Ocean). The F_t and E_t terms were input to the model. Estimated parameters in eq. A9 were STEP$_j$, which was equal to $-\ln(\lambda_n)$ in the Delta model (see eq. A5) and estimated for brood years after 1974 (assumed to be zero before 1975), and the coefficients c_1 and c_2. The Alpha version of the model thus estimated an estuary and ocean survival factor (STEP) directly as a stepped increase in estuary and ocean mortality after brood year 1974 and then used STEP in conjunction with passage model estimates of passage-related survival and estimated relationships between estuary and ocean survival and various climatic factors to calculate an overall spawner-to-recruit survival rate.

Note that although the Ricker a_i term in the Alpha model was defined and estimated in a similar manner as in the Delta model, it was not directly comparable because of the subtraction of averages in the $\alpha_{t,j}$ term (eq. A8). Adjustment of the Alpha model Ricker a_i term by addition of averages in the $\alpha_{t,j}$ term was necessary to make the Alpha and Delta model Ricker a_i terms comparable.

Comparison of the Delta and Alpha versions

With the definitions below, both the Delta and Alpha versions of the model can be written as a Ricker model with four specially defined terms:

$$\text{(A10)}\quad \ln(R) = (1 + p)\ln(S) + \text{Term1} + \text{Term2} + \text{Term3} + \text{Term4} - bS + \varepsilon$$

where Term1 was the Ricker a value, representing the collection of all constant terms in each model other than those needed to center climate variables modeled as Markov processes. In the Delta model, this was the term "a"; in the Alpha model, this was the term "a – average($\ln(DP_{bt} + 1 - P_{bt})$) + average(STEP)". Term2 was the logarithm of system survival, $\ln(\omega)$. In both models, this was the term "$-M + \ln(DP_{bt} + 1 - P_{bt})$" and was supplied by the passage and transportation models. Term3 was the logarithm of post-Bonneville survival factor of in-river fish, $\ln(\lambda_n)$. In the Delta model, this was the term "$-m - \ln(\omega)$"; in the Alpha model, this was the term "–STEP". Term 4 was represented by Markov-type climate variables, centered so that they sum to zero over brood years 1951–1990. In the Delta model, this was the term δ_t; in the Alpha model, this was the term "$c_1(1/F - \text{average}(1/F)) + c_2(E/F - \text{average}(E/F))$".

Hypotheses for estuary and ocean survival factor for in-river fish (λ_n)

Hydro hypothesis

The Hydro hypothesis postulated that λ_n has been, and will be, related to delayed effects of passage (such as increased stress) through the eight dams of the Federal Columbia River Power System. Evidence for this hypothesis is reviewed in Budy et al. (2002). The Hydro hypothesis was implemented in the life-cycle model by selecting a particular flow year (from 1977 to 1992) based on the frequency of that type of year in the historical (1929–1992) record and then applying eq. A11.

$$\text{(A11)}\quad (1 - \lambda_{n,p}) = (1 - \lambda_{n,r}) \cdot \frac{(1 - V_{n,p})}{(1 - V_{n,r})}$$

where $\lambda_{n,p}$ is the prospective estuary and ocean survival factor for in-river fish for a future year *p*, a future year with unregulated flow equal to that of a historical flow year *r*. $\lambda_{n,p}$ was estimated from $\lambda_{n,r}$ (retrospective estuary and ocean survival factor for in-river fish for historical flow year *r*, calculated in retrospective life-cycle modeling), $V_{n,p}$ (prospective passage survival rate of in-river fish for future flow year *p*, calculated in prospective passage modeling), and $V_{n,r}$ (retrospective passage survival rate of in-river fish for historical flow year *r*, calculated in retrospective passage modeling).

Stock viability hypothesis

The stock viability hypothesis proposed that estimates of λ_n for the 1977–1992 period will apply indefinitely in the future, even if hydrosystem direct mortality declines or climate improves. Several mechanisms to account for decreased stock viability were proposed, including an increase in incidence or severity of bacterial kidney disease (BKD) in wild populations resulting from increased numbers of hatchery fish, increased predation rates on juveniles because of low stock sizes, and insufficient nutrients from returning adults' carcasses to support the growth of parr. The key feature of this hypothesis was that future hydrosystem actions, even if they improved survival rates through the hydropower system, would have no effect on estuary and ocean survival rates of in-river fish. The stock viability hypothesis was implemented in the life-cycle model by selecting (in each simulation and year) a particular flow year (from 1977 to 1992) based on the frequency of that type of year in the historical (1929–1992) record and then setting the prospective value of λ_n equal to the retrospective value for that flow year.

Regime shift hypothesis

The regime shift hypothesis suggested that the estuary and ocean survival factor for in-river fish was due to a long-term (60-year) cycle in ocean climate. This regime is believed to have shifted from good to poor during brood year 1975 and is expected to return to above-average conditions in 2005 (Beamish et al. 1999). Key assumptions in this hypothesis were that cyclical changes in climate affect salmon production (Hare et al. 1999) and that these climatic effects on Snake River stocks are systematically different from effects on lower Columbia River stocks.

In the life-cycle model, the regime shift hypothesis was implemented by selecting a flow year from the historical period according to the phase of the 60-year climatic cycle. For forward simulation years up to 2005, flow years were selected from the "bad" phase of the cycle (1975–1990). From simulation years 2006 to 2035, flow years were selected from a period of good ocean conditions (1952–1974). For each selected flow year, the prospective value of λ_n was assumed to be the same as the retrospective value estimated for that flow year.