

# Application of decision analysis to evaluate recovery actions for threatened Snake River fall chinook salmon (*Oncorhynchus tshawytscha*)

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**Abstract:** We used models and decision analysis to incorporate uncertainties into evaluations of two recovery actions for Snake River fall chinook salmon (*Oncorhynchus tshawytscha*): (i) maximize transportation and (ii) natural river drawdown of four lower Snake River dams. In the retrospective analysis, we compared alternative stock–recruit models and selected one that was consistent with historical spawner–recruit data and allowed us to implement alternative hypotheses about hydrosystem, hatchery, harvest, and environmental effects. In the prospective analysis, we used this model and posterior distributions of its parameters in a decision analysis framework to compare projected escapements for the two actions over a range of alternative hypotheses. We found that drawdown was most risk averse, producing larger long-term escapements than maximizing transportation under most hypotheses and model assumptions. Maximizing transportation and drawdown produced similar escapements only if we assumed high or increasing estuary and ocean survival rates of transported fish coupled with either severe reductions in harvest rates or insensitivity of upstream survival rates to dam construction and removal. Although there was relatively little information available for Snake River fall chinook (particularly about estuary and ocean survival rates of transported smolts), decision analysis was a useful technique for organizing data, assessing actions over a range of uncertainties, and identifying research priorities.

**Résumé :** Des modèles et une analyse décisionnelle nous ont permis d'incorporer les incertitudes dans l'évaluation de deux stratégies de récupération des populations automnales des Saumons quinnat (*Oncorhynchus tshawytscha*) de la rivière Snake, soit de maximiser le transport des saumoneaux, soit de faire baisser les eaux de la rivière à leur niveau naturel à quatre barrages du bassin inférieur. Dans une analyse rétrospective, nous avons comparé divers modèles de stock–recrutement de rechange et en avons choisi un qui correspondait aux données historiques sur les géniteurs et les recrues et qui nous permettait d'avancer diverses hypothèses sur les effets de l'hydrosystème, de la pisciculture, de la récolte et de l'environnement. Dans notre analyse prospective, ce modèle et les distributions a posteriori de ses paramètres nous ont permis de comparer, dans le cadre d'une analyse décisionnelle, les échappées prédites pour chacune des stratégies dans une gamme d'hypothèses de rechange. Nous avons trouvé que la baisse du niveau était la stratégie qui posait le moins de risques; elle produisait aussi à long terme des échappées plus fortes que l'augmentation du transport sous la plupart des hypothèses et des présuppositions du modèle. La maximisation du transport et la baisse du niveau donnaient des échappées semblables seulement si on présupposait que les poissons transportés avaient une survie élevée ou plus grande dans l'estuaire et l'océan; il fallait aussi qu'il y ait une réduction importante des taux de récolte ou alors que les taux de survie en amont ne soient pas affectés par la construction ou le retrait des barrages. Bien qu'il y ait eu relativement peu de renseignements disponibles sur la population automnale des Saumons quinnat de la rivière Snake (en particulier sur les taux de survie en estuaire et en océan des saumoneaux transportés), l'analyse décisionnelle s'est avérée utile pour organiser les données, évaluer les stratégies dans une gamme d'incertitudes et identifier les priorités de recherche.

[Traduit par la Rédaction]

## Introduction

The Plan for Analyzing and Testing Hypotheses (PATH) was a multiagency research program to resolve biological uncertainties about recovery of chinook salmon (*Oncorhynchus tshawytscha*) and steelhead (*Oncorhynchus mykiss*)

stocks in the Snake River sub-basin of the Columbia River (Marmorek and Peters 2001). PATH grew out of a review of three modeling systems (Barnhouse 1993) and a subsequent assessment by the National Marine Fisheries Service (NMFS 1995). Both documents recommended examining the models' underlying hypotheses. PATH's objectives were to ex-

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.  
J15925

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amine these hypotheses in a rigorous but collaborative manner, resolve as many differences as possible, and evaluate alternative actions in the light of remaining uncertainties.

There were two phases of PATH analyses. "Retrospective analyses" formulated hypotheses and assessed their consistency with historical data sets. Available data were sufficient to resolve only some uncertainties about the effects of alternative actions and environmental factors on survival rates. Remaining uncertainties were carried forward in "prospective analyses", which used models of Snake River salmon to project escapements resulting from alternative recovery actions. We used decision analysis as a framework for systematically incorporating uncertainties into the evaluation of alternative recovery actions (Peterman and Anderson 1999). This provided a mechanism for determining the relative influence of various sources of uncertainty on the outcomes of alternative actions and identifying risk-averse recovery actions that performed well over a range of uncertainties. PATH completed retrospective and prospective analyses of Snake River spring- and summer-run chinook (Deriso et al. 2001; Peters and Marmorek 2001) and ocean type (juveniles spend <1 year in freshwater) fall-run chinook (this study).

Analyses of fall chinook were less detailed than spring and summer chinook because there were less data. For spring and summer chinook, we had 33–38 years of spawner–recruit data for seven Snake River and three lower Columbia River stocks, numerous transportation studies conducted between 1968 and 1995, and 19 years of juvenile and smolt-to-adult survival rate estimates between 1966 and 1996. In contrast, for fall chinook, we had 20–27 years of spawner–recruit data for only three stocks and 4 years of juvenile survival data (all from recent years). In addition, sample sizes have been too small to produce reliable data for evaluating the benefits of transportation for Snake River fall chinook. The lack of data for fall chinook tested the utility of our decision analysis methods under conditions of high uncertainty.

## Methods

### Description of stocks

PATH reconstructed spawner and recruit information for three fall chinook stocks in the Columbia Basin (Fig. 1; Table 1). The Snake River population (the focus of this study) consists of adult fall chinook spawning between Hells Canyon and Lower Granite dams. The proportion of spawners of hatchery origin (hatchery fraction) has been substantial since 1990. After rearing in the mainstem of the Snake River, juveniles migrate through eight dams on the Snake and Columbia rivers or are collected at upper dams and transported by truck to below Bonneville Dam. Hatchery fractions, survival rates through the hydrosystem, and proportion transported are summarized in Table 1. Adult fish experience ocean and in-river harvest rates of around 30% and interdam losses during upstream migration of 40–50%.

### Retrospective analysis

The purpose of the retrospective analysis was to identify a model structure for prospective analyses, a set of hypotheses to be carried forward into the prospective analyses, and a distribution of parameter values consistent with the spawner–recruit data. Spawner–recruit survival anomalies for Snake River fall chinook (residuals from predicted recruits per spawner using a standard Ricker (1975) spawner–recruit model) declined starting in the early 1980s

(Fig. 2). This period was characterized by full operation of the Snake River dams, increasing transportation of smolts, and increased proportion of hatchery fish in the spawning population (Peters et al. 1999).

We explored the effects of these and other mortality factors on observed survival rates of Snake River fall chinook using passage and life-cycle models (Fig. 3). Passage models computed the total direct in-river and transport mortality ( $M$ ) of Snake River smolts through the eight dams of the Federal Columbia River Power System and the proportion of smolts arriving below Bonneville Dam that were transported ( $P_{bt}$ ). Values were generated for brood years 1964 to 1991 (Table 1), based on assumptions about migration timing, survival rates in free-flowing reaches, predation rates, bypass, turbine, spillway, and transport mortality, temperature effects, fish travel time, and effectiveness of predator-removal programs (see Peters et al. (1999) for details).  $M$  and  $P_{bt}$  were assumed to be calculated without error.

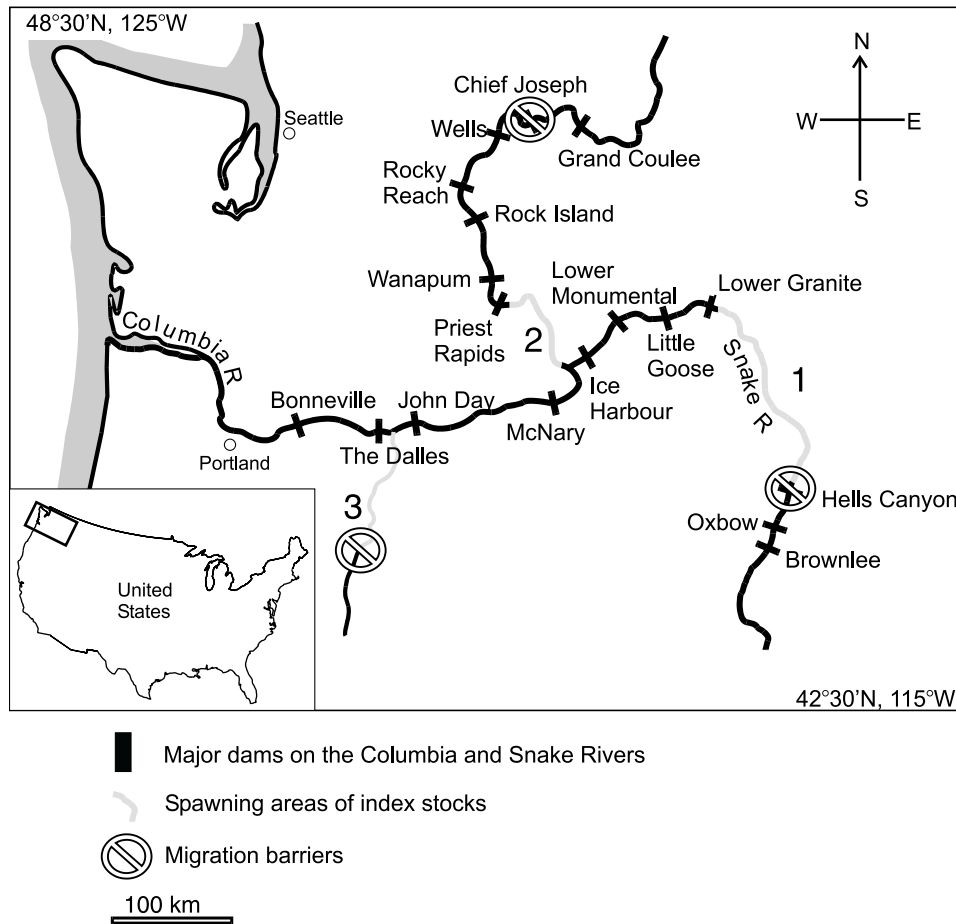
Survival of Snake River fall chinook through the spawning, rearing, and adult life stages was modeled with a generalized Ricker spawner–recruit model:

$$(1) \quad R_t = (S_{wild,t} + E \cdot S_{other,t})^{(1+p)} e^{\alpha} e^{-b(S_{wild,t} + E \cdot S_{other,t})} e^{-M_t} \\ \times (DP_{bt} + 1 - P_{bt}) e^{\text{clim}} e^{-\text{STEP}} e^{e_t}$$

where  $M_t$  and  $P_{bt}$  were computed by passage models, and  $R_t$  (number of returns to the Columbia River in the absence of ocean harvest),  $S_{wild,t}$  (number of wild age-3+ spawners in year  $t$ ), and  $S_{other,t}$  (number of hatchery spawners in year  $t$ ) were obtained from run reconstructions (Table 1).  $S_{wild,t}$  and  $S_{other,t}$  were assumed to be measured without error, based on the insensitivity of PATH's spring and summer chinook model to spawner measurement error (Deriso et al. 2001). Estimated parameters in eq. 1 were the Ricker  $\alpha$  and  $b$  parameters, a depensation parameter ( $p$ ) to reflect potential changes in survival at low spawner abundance, a climate factor accounting for mortality sources other than those specified by other parameters in the model (clim), and overall residual variation in the Snake River stock in year  $t$  ( $e_t$ ). Both measurement error in the returns ( $R_t$ ) and process error (natural variation) contributed to the error term  $e_t$ .  $D$  was the ratio of post-Bonneville (estuary and ocean) survival rate of transported smolts to post-Bonneville survival rate of nontransported smolts (i.e., if  $D < 1$ , transported fish have lower post-Bonneville survival rates than nontransported fish).  $D$  was assumed to be a constant during all years of transportation and was either specified or estimated. STEP represented the average increase in overall mortality after accounting for the hypothesized delayed effects of transportation ( $D$ ) and total direct passage mortality ( $M_t$ ). This increase in overall mortality occurred in either brood year 1970 (presumed to be related to the construction and start-up of the four lower Snake River dams) or in 1976 (presumed to be related to changes in ocean conditions or to the start of full operation of the four lower Snake River dams). We also estimated a parameter to represent relative spawning effectiveness of hatchery fish ( $E$ ), ranging from 0 (hatchery spawners made no contribution to future generations) to 1 (hatchery spawners made the same contribution as wild fish).

We explored 30 variations in the structure of eq. 1 representing alternative hypotheses about the effects of various mortality factors, and evaluated their consistency with the historical spawner–recruit data (Table 2). All 30 variations omitted the depensation parameter  $p$  because we found that the estimated value of this parameter was always close to zero, indicating no evidence for depensatory effects in the spawner–recruit data. Variations varied in (i) complexity (from simple two-parameter Ricker models to complex Ricker models such as eq. 1); (ii) passage models (either CRiSP (Columbia River Salmon Passage), developed by researchers at the University of Washington, or Fall FLUSH (Fish Leaving Under Several Hypotheses), developed by scientists at State and

**Fig. 1.** Location of fall chinook index populations: 1 = Snake River (fish pass through eight mainstem dams); 2 = Hanford Reach (4 dams); 3 = Deschutes River (2 dams). Approximate latitude and longitude are given in the corner of the diagram.



Tribal fisheries agencies); and (iii) years in which STEP takes effect (either 1970 or 1976). Some models estimated  $D$  and  $E$ , whereas others treated these parameters as fixed inputs. The model variations also included several indirect approaches for estimating annual climate factors because there were insufficient degrees of freedom in the spawner–recruit data to estimate these directly. Approaches ranged from omitting the climate factor entirely to using residuals from a Ricker model fit to the Hanford or Deschutes fall chinook data. In models that used Deschutes residuals, we used the significant positive correlation between Deschutes and Snake residuals ( $r = 0.75$ ) to reduce the residual error in eq. 1:

$$(2) \quad e_t = g \cdot e_t(\text{des}) + e'_t$$

where  $e_t(\text{des})$  represents residual variation in Deschutes stock in year  $t$  (estimated from fitting a standard Ricker stock–recruit model to Deschutes spawner–recruit data),  $g$  is the covariation coefficient (estimated), and  $e'_t$  is the remaining residual variation in Snake River stock after correlation with Deschutes stock has been removed. This assumed that  $e_t(\text{des})$  and  $e'_t$  were independent normal random variables. In models that adopted this structure, the climate factor in eq. 1 (clim) dropped out and was absorbed into the  $e_t$  term.

We first compared the 30 models to spawner–recruit data and then estimated a distribution of parameter values for selected models. Both the model comparison and parameter estimation analyses were based on the unnormalized posterior density function:

$$(3) \quad \Pr\{e_t(\text{des})\} \cdot \Pr\{e'_t\} \cdot \Pr\{\text{priors}\}$$

where the first two terms represent the likelihood functions associated with the Deschutes and Snake River recruitment data, respectively (Gelman et al. 1995; Quinn and Deriso 1999). This involved simultaneous fitting of Ricker-type models to the Deschutes and Snake River stocks. The last term represented the prior distribution for the model parameters. We assumed uniform prior distributions for all parameters, except for the Snake River Ricker  $b$  coefficient. For this parameter, we assumed a prior distribution

$$(4) \quad \ln(b) \sim N(\ln(1/5000), 0.25)$$

based on an independent analysis that suggested recruitment was maximized at 5000 spawners (Schaller and Cooney 1992; see McAllister and Kirkwood (1998) for a review of the use of priors in fisheries). Model fits to spawner–recruit data were insensitive to this assumption.

To compare the models to spawner–recruit data, we first calculated the mode of eq. 3 for each model by maximizing that function with respect to unknown parameters. Likelihoods for each of the alternative models were then compared using the Akaike Information Criterion (AIC) and Bayes Information Criterion (BIC), which are essentially penalized versions of the likelihood ratio method (Kass and Raftery 1995; Fournier et al. 1998). Models with lower AIC and BIC scores fit the data better than models with higher scores, and as a general guideline, we considered two models to be “significantly” different when the difference in their AIC or BIC scores was  $>2$ . For example, the fit of model 1 (AIC = 49.03) was deemed to be significantly different from model 2 (AIC = 59.48) because the difference in their AIC scores was  $>2$ , and the lower AIC score of model 1 suggested a better fit to the

**Table 1.** Spawner (*S*) and recruit (*R*) data for Snake River, Hanford Reach, and Deschutes River fall chinook stocks, brood years 1964–1999 (Deschutes data start in brood year 1977).

Brood year	Snake River					Hanford Reach		Deschutes River	
	<i>S</i>	<i>R</i>	$H_{frac}$	$e^{-M}$	$P_{bt}$	<i>S</i>	<i>R</i>	<i>S</i>	<i>R</i>
1964	7 648	35 240	0.00	0.47	0.00	22 703	100 043	—	—
1965	6 339	62 471	0.00	0.43	0.00	26 668	239 681	—	—
1966	8 623	34 329	0.00	0.46	0.00	29 724	193 231	—	—
1967	10 414	71 436	0.00	0.38	0.00	24 638	307 471	—	—
1968	17 556	48 681	0.00	0.31	0.00	24 035	263 670	—	—
1969	4 649	35 129	0.00	0.25	0.00	28 937	286 328	—	—
1970	4 353	43 363	0.00	0.27	0.00	20 511	590 130	—	—
1971	4 091	22 699	0.00	0.27	0.00	26 393	471 622	—	—
1972	1 371	17 390	0.00	0.20	0.00	19 327	361 190	—	—
1973	2 194	15 716	0.00	0.27	0.00	36 343	398 212	—	—
1974	668	12 910	0.00	0.21	0.00	28 940	333 580	—	—
1975	1 387	10 619	0.00	0.22	0.00	34 628	268 136	—	—
1976	691	7 019	0.00	0.16	0.23	39 987	108 581	—	—
1977	1 011	9 259	0.00	0.22	0.21	40 745	107 827	6414	17 641
1978	841	4 946	0.00	0.20	0.33	21 644	56 563	4099	16 172
1979	802	11 657	0.00	0.26	0.45	24 840	164 027	3728	15 831
1980	515	7 817	0.00	0.37	0.69	21 224	304 686	2788	15 490
1981	878	4 746	0.00	0.35	0.58	14 213	265 436	4704	17 145
1982	1 209	7 500	0.00	0.37	0.67	22 598	458 905	5176	15 725
1983	909	8 723	0.07	0.39	0.65	37 038	647 038	4160	16 090
1984	717	9 721	0.23	0.37	0.76	48 149	956 878	2690	56 348
1985	1 080	4 821	0.18	0.35	0.74	71 732	274 308	6333	11 974
1986	1 403	4 971	0.24	0.35	0.81	100 626	239 529	6045	11 576
1987	1 064	2 171	0.57	0.37	0.80	105 347	101 086	6278	4 125
1988	702	3 748	0.29	0.34	0.72	96 329	96 391	7903	8 804
1989	815	2 031	0.49	0.39	0.79	72 022	151 284	3927	10 043
1990	273	975	0.77	0.67	0.93	47 856	131 271	2320	14 416
1991	767	717	0.34	0.66	0.96	37 580	38 067	3684	5 765
1992	674	—	0.17	—	—	34 371	—	3454	—
1993	883	—	0.21	—	—	35 322	—	6126	—
1994	448	—	0.32	—	—	54 373	—	6025	—
1995	226	—	0.43	—	—	39 936	—	6603	—
1996	964	—	0.35	—	—	38 443	—	7734	—

**Note:** Spawner estimates are the total of natural and hatchery-origin spawners. Recruits are the number of adults at the mouth of the Columbia River that would have been produced by the spawners in a particular brood year in the absence of ocean harvest. Recruits were back-calculated from spawner data using historical estimates of upstream survival rates, in-river harvest rates, and maturity schedules.  $H_{frac}$ , fraction of spawners that were of hatchery origin;  $e^{-M}$ , weighted average survival rate of transported and in-river fish through the hydropower system;  $P_{bt}$ , fraction of fish below Bonneville Dam that were transported.  $P_{bt}$  and  $e^{-M}$  are the average of CRiSP (Columbia River Salmon Passage) and fall FLUSH (Fish Leaving Under Several Hypotheses) passage model estimates.

data than that of model 2. We then numerically sampled parameters in our selected models using a Monte Carlo Markov Chain algorithm (Gelman et al. 1995) to get parameter distributions for the prospective analysis. Further Monte Carlo analyses revealed only a small bias in our estimates of Ricker *a*, Ricker *b*, *p*, *D*, and *E* (Hinrichsen 2001).

## Prospective and decision analysis

### Alternative actions to be evaluated

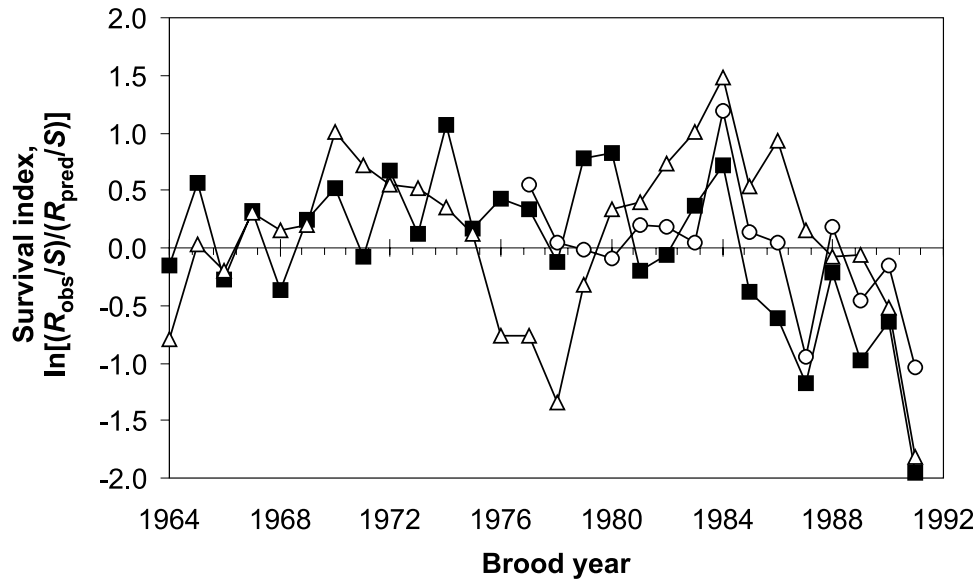
The decision analysis focused on two recovery actions related to the operation of the Columbia River hydropower system. In action A2 (or “maximize transportation”), the system is operated to collect as many juvenile fish as possible at the uppermost dams for transport by truck to below Bonneville Dam. This was equivalent to a “status quo” action because the system is currently operated to maximize transportation of fall chinook smolts. Action A3 was natural river drawdown of the four dams on the lower Snake River.

With this action, passage around these dams would be restored by removing earthen berms adjacent to the dam structures. Navigation, power generation, and collection of smolts for transportation would be eliminated at these projects.

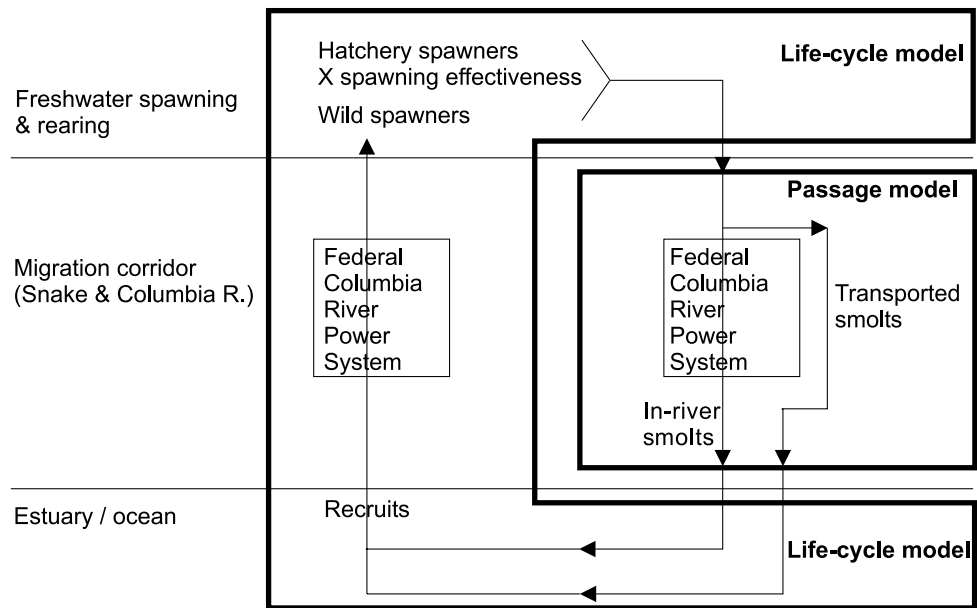
### Performance measures to evaluate the actions

PATH used the jeopardy standards largely accepted by NMFS for Snake River chinook salmon (NMFS 1995) as our primary performance measure (see Peters and Marmorek (2001) for details). The standards were based on the probability (calculated over multiple Monte Carlo simulations) that projected escapement of natural origin spawners would exceed survival and recovery thresholds, which for Snake River fall chinook were 300 and 2500 spawners, respectively. Probabilities of exceeding the survival threshold of 300 spawners were calculated over simulation years 1–24 and 1–100 to capture short- and long-term effects; the probability of exceeding the recovery threshold was calculated over simulation years 41–48.

**Fig. 2.** Survival anomaly index for Snake River (solid squares), Hanford Reach (open triangles), and Deschutes River (open circles) fall chinook. Annual survival anomalies are the residuals from  $\ln(R/S)$  predicted from the estimated Ricker stock–recruit curve. Negative values indicate lower survival rates than the average over the entire period.



**Fig. 3.** Schematic of the life cycle of Snake River fall chinook salmon. Passage models estimated survival of smolts in the Columbia and Snake rivers as they pass through (in-river smolts) or around (transported smolts) the Federal Columbia River Power System (FCRPS). Life-cycle models estimated survival rates over the rest of the life cycle.



*Models to project outcomes of actions*

The prospective modeling framework included hydro-regulation, passage, and life-cycle models (Fig. 4). The U.S. Army Corps of Engineers’ HYSER hydro-regulation model translated each action into 18 sets of mean monthly flows in the Snake and Columbia rivers (one for each year from 1976 to 1993). Passage models projected annual values for the two passage-related parameters ( $M$ ,  $P_{bt}$ ) for each action and for each of the 18 sets of flows that were expected from that action. For example, passage-related parameters generated from the 1988 mean monthly flows represented the expected passage-related survival rates if a given action were implemented and if 1988 natural flow conditions occurred.

The prospective life-cycle model projected the effects of future alternative management actions on spawning escapement of Snake River fall chinook using posterior distributions of model parameters from the retrospective modeling (i.e., Ricker  $a$  and  $b$ ,  $E$ ,  $g$ , climate factor, and either  $D$  or STEP) and assumptions about passage survival, ocean and in-river harvest, upstream survival rates, hatchery supplementation, and carrying capacity of spawning habitat under drawdown (Peters et al. 1999). The life-cycle model performed 4000 Monte Carlo simulations of spawning abundance of the Snake River fall chinook stock over 100 years, beginning with brood year 1997. Each simulation drew from the posterior distribution of parameters, as well as a particular “flow year” from 1976 to

**Table 2.** Combinations of model assumptions compared in retrospective analysis of life-cycle models.

Model	Passage model	Climate factor	<i>E</i>	<i>D</i>	STEP year	STEP	No. of pars	AIC	BIC
1	CRiSP	Deschutes–Snake <sup>a</sup>	0.77	0.03	1970	0.02	6	49.03	57.02
2	CRiSP	None	1.00	0.00	1970	0.00	5	59.48	66.14
3	CRiSP	None	0.13	0.14*	1970	0.08	4	61.52	66.85
4	CRiSP	None	0.00	0.37*	1970	0.00	4	68.72	74.05
5	CRiSP	Ocean Temp. <sup>b</sup>	0.64	0.00	1970	0.00	6	59.64	67.64
6	CRiSP	Spring–summer $\delta^c$	0.55	0.01	1970	0.00	6	59.55	67.54
7	CRiSP	PAPA index <sup>d</sup>	0.58	0.00	1970	0.00	6	60.43	68.42
8	CRiSP	Hanford <sup>e</sup>	0.55	0.03	1970	0.16	6	55.76	63.75
9	CRiSP	Deschutes–Snake	0.17	0.37*	1976	0.37	5	55.02	61.68
10	CRiSP	Deschutes–Snake	0.00*	0.17	1970	0.09	5	54.07	60.73
11	CRiSP	Deschutes–Snake	0.70*	0.03	1970	0.02	5	47.08	53.75
12	CRiSP	Deschutes–Snake	1.00*	0.02	1970	0.03	5	47.17	53.84
13	FLUSH	Deschutes–Snake	0.87	0.03	1970	0.00	6	56.21	64.21
14	FLUSH	None	1.00	0.00	1970	0.00	5	62.81	69.47
15	FLUSH	None	0.06	0.14*	1970	0.00	4	66.29	71.62
16	FLUSH	None	0.00	0.37*	1970	0.00	4	74.24	79.56
17	FLUSH	Ocean Temp.	0.85	0.00	1970	0.00	6	64.45	72.44
18	FLUSH	Spring–Summer	0.82	0.00	1970	0.00	6	64.35	72.34
19	FLUSH	PAPA index	0.73	0.00	1970	0.00	6	64.10	72.09
20	FLUSH	Hanford	0.64	0.02	1970	0.00	6	62.55	70.54
21	FLUSH	Deschutes–Snake	0.08	0.37*	1976	0.35	5	63.16	69.82
22	FLUSH	Deschutes–Snake	0.00*	0.12	1970	0.00	5	59.47	66.13
23	FLUSH	Deschutes–Snake	0.70*	0.04	1970	0.00	5	54.32	60.98
24	FLUSH	Deschutes–Snake	1.00*	0.02	1970	0.00	5	54.24	60.90
25	None	None	1.00*	NA	NA	No STEP	2	73.63	76.30
26	None	Deschutes–Snake	0.12	NA	NA	No STEP	4	51.21	56.54
27	None	Deschutes–Snake	0.14	NA	1970	-0.47	5	53.18	59.84
28	None	Deschutes–Snake	0.24	NA	1976	-0.58	5	49.40	56.00
29	CRiSP WTT <sup>f</sup>	Deschutes–Snake	0.64	NA	NA	No STEP	6	47.29	55.29
30	FLUSH WTT <sup>f</sup>	Deschutes–Snake	0.67	NA	NA	No STEP	6	46.31	54.31

**Note:** *E*, spawning effectiveness of hatchery fish; *D*, ratio of post-Bonneville survival of transported fish to post-Bonneville survival of in-river (nontransported) fish; \* after number in *E* and *D* columns indicates that value is fixed; pars, Snake River parameters estimated. AIC (Akaike Information Criterion) and BIC (Bayes Information Criterion) are measures of the fit of the model to the historical spawner–recruit data. Lower AIC and BIC scores indicate better fits to the data; differences in AIC and BIC > 2 represent significant differences in model fit. CRiSP, Columbia River Salmon Passage; FLUSH, Fish Leaving Under Several Hypotheses; WTT, water travel time, the seasonal average number of days that it takes water to flow from Lower Granite Dam to Bonneville Dam; NA, not applicable.

<sup>a</sup>Joint Snake and Deschutes residuals resulting from simultaneous fitting of Ricker stock–recruit models to both stocks.

<sup>b</sup>Ocean temperature index (Hyun 1996) based on average sea surface temperature during the period of October–January at five coastal Canadian weather stations (Langara Is., Cape St. James, Pine Island, Kains Is., and Amphitrite Point).

<sup>c</sup>Maximum likelihood estimates of year-effects ( $\delta$ ) from the spring–summer chinook data (Deriso et al. 2001).

<sup>d</sup>Latitude of a drifting object 3 months after release from Ocean Station PAPA in the Northeast Pacific Ocean.

<sup>e</sup>Residuals from a simple two-parameter Ricker model fit to Hanford spawner–recruit data.

<sup>f</sup>These models replaced CRiSP and FLUSH estimates of *M* and *P<sub>bt</sub>* with passage model estimates of the proportion of fish arriving at Lower Granite Dam that were transported and the water travel time (WTT).

1993 based on the representation of those years in the historical record (1929–1993). The selected flow year determined which of the 18 sets of passage-related parameters were used in that simulation year.

#### *Uncertainties that determine the range of possible outcomes for each action*

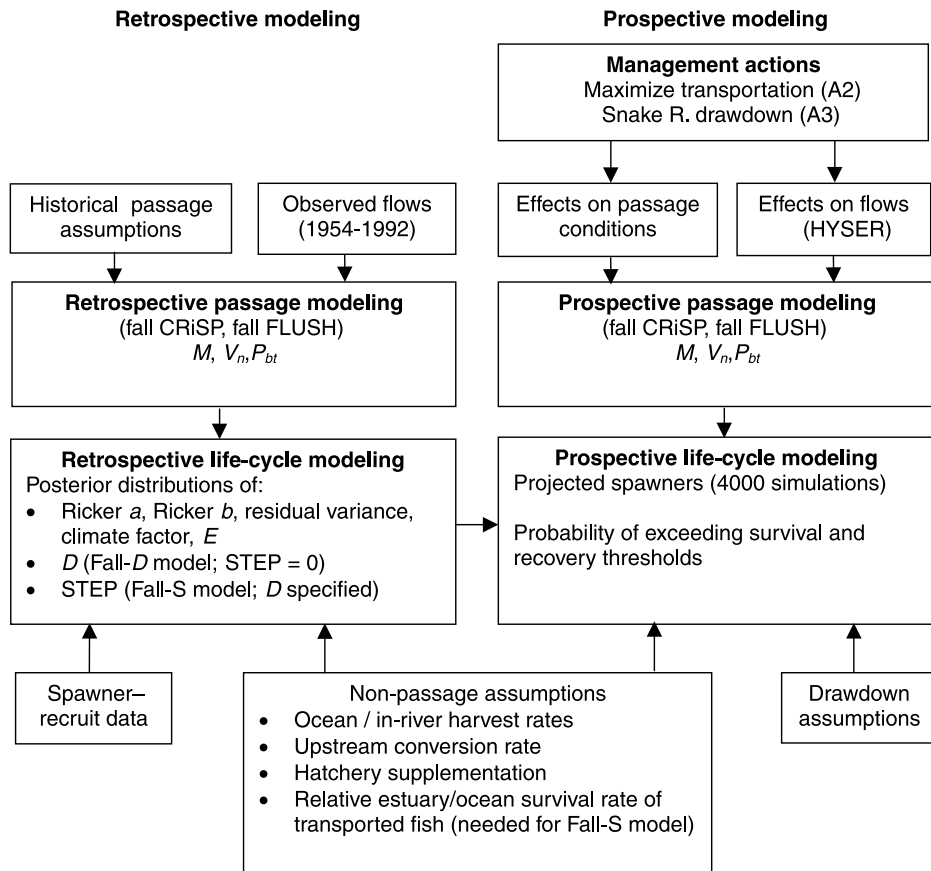
We included two types of uncertainties in the decision analysis framework. Uncertainty in life-cycle model parameters was included through Monte Carlo simulations (described above). PATH scientists also included discrete alternative hypotheses for six other uncertainties (Table 3; Peters et al. 1999). These uncertainties were selected either because they were important in the spring and summer chinook decision analysis, because the retrospective analysis of fall chinook revealed considerable uncertainty, or because they

were related to future conditions and thus not addressed by retrospective analyses.

*Juvenile passage survival*—Assumptions about factors affecting the survival rate of smolts through the hydropower system (e.g., effects of temperature on mortality rate, the length of time that it takes fish to travel through the hydrosystem, partitioning of in-river survival between dam and reservoir survival, and the proportion of smolts transported) were embodied in the two passage models, fall CRiSP and fall FLUSH. The passage models generated deterministic predictions of total direct passage mortality (*M*) and proportion transported (*P<sub>bt</sub>*).

*Relative post-Bonneville survival of transported smolts*—The relative survival rates of transported smolts in the estuary and ocean (*D*) are generally estimated from transportation studies which com-

**Fig. 4.** Modeling approach used in the decision analysis. Retrospective modeling (left side) described historical patterns in spawner-to-recruit survival rates and estimated distributions of parameters, which were passed to the prospective analysis (right side of diagram). Prospective analyses combined retrospective results with future assumptions to forecast spawning escapements under a range of actions, assumptions, and hypotheses. See text for definition of terms.



pare smolt-to-adult survival rates (SAR) of transported and non-transported groups of PIT-tagged (passive induced transponder) fish. Past and current *D* values for Snake River fall chinook were highly uncertain because there were only 2 years of limited SAR data (1995 and 1996) and because transport methods have changed between the retrospective (brood years 1964–1991) and prospective (post-1991) periods. We implemented four *D* hypotheses, each consisting of a hypothesized retrospective and prospective *D* value.

**Post-Bonneville mortality (STEP)**—STEP was assumed to be zero prior to 1976; afterwards, it was assumed to take on the value of the estimated change in productivity not accounted for by the total direct passage mortality (*M*) and the effectiveness of transportation (*D*) from brood years 1976–1991. The interaction between *D* and STEP led to two model forms for use in prospective modeling. In one version (“Fall-*D* model”), *D* was estimated from the spawner–recruit data (as in hypotheses D1 and D3 in Table 3) and STEP was set to zero. In the other form (“Fall-*S*”), *D* was specified as a fixed input value and STEP was estimated. In the Fall-*S* model, we implemented three alternative hypotheses about how estimated STEP values were applied in future simulations. The rationale for the two model forms is described more fully in the retrospective results section below.

**Duration of pre-removal period (drawdown action only)**—There may be delays in initiating drawdown once such a decision is made because of the Congressional appropriations process and the possibility of litigation.

**Juvenile survival rate after drawdown (drawdown action only)**—We considered two hypotheses about juvenile survival rates in the 222-km unimpounded reach that would be created by drawdown, based on two analyses of survival rate data collected in a free-flowing reach of the Snake River above Lower Granite Reservoir in 1995, 1997, and 1998. These analyses suggested post-drawdown survival rates of juveniles through the Snake and Columbia rivers (with only four dams remaining) of 0.3 to 0.51, depending on the assumptions used in the analysis and the passage model used to calculate survival through the four remaining dams. Current in-river survival rates through the existing eight dams averaged 0.29 with CRiSP and 0.22 with FLUSH.

**Duration of transition period (drawdown action only)**—The transition period represented the amount of time that it would take for the survival rate of juvenile fall chinook to achieve dynamic equilibrium following drawdown. The length of this period depended on assumptions about how long it would take for prey and predator communities to re-establish after drawdown.

**Probabilities on alternative hypotheses**

Probabilities on alternative hypotheses reflect their relative likelihood or strength of empirical support. PATH did not complete a Weight of Evidence process for fall chinook as we did for spring and summer chinook (Marmorek and Peters 2001), so we simply assumed that all hypotheses were equally likely and conducted sensitivity analyses to determine which of the uncertainties had the greatest effect on outcomes.

**Table 3.** Definition of alternative assumptions addressed in the analyses.

Model component	Alternative assumptions
Total passage mortality ( $M$ ) and proportion transported ( $P_{bt}$ )	CRiSP estimates Fall FLUSH estimates
Relative post-Bonneville survival of transported fish outside of the hydrosystem ( $D$ )	D1: retrospective $D$ values estimated from spawner–recruit data using eq. 1 with STEP set = 0 (maximum likelihood estimate of $D \approx 0.05$ ). Prospectively, $D$ increases to 0.24 (mean value estimated from 1995 PIT-tag data), because starting in brood year 1992, smolts were released midchannel rather than along shore (where large concentrations of predators were observed). D2: Retrospective and prospective $D$ values = 1.0 (based on transportation studies of Hanford Reach fall chinook conducted at McNary dam in 1978–1983). $D$ values for Hanford fish may not apply to Snake River fish because of differences in genetics, spatial distribution, smolt size, and outmigration timing. D3: Retrospective and prospective $D$ values estimated from spawner–recruit data using eq. 1. This approach maximized fit of the life-cycle model to spawner–recruit data but assumed that historical estimates were applicable in prospective scenarios despite changes in transportation methods after 1991 (the last year of data used to estimate $D$ ). D4: Retrospective and prospective $D = 0.2$ (mean estimate from 1995 and 1996 PIT-tag data).
Post-Bonneville mortality (STEP)	Hydro: future STEP = value selected from posterior distribution, unless the Snake River dams are breached (action A3), in which case STEP = 0. Assumed that STEP increase in mortality in brood year 1976 was the result of delayed effects of passing through or around the four lower Snake River dams, which were in full operation by that time. Stock viability: future STEP = a value selected from posterior distribution, even if Snake River dams are breached. Assumed that 1976 STEP increase in mortality was the result of factors unrelated to hydrosystem actions or climate (e.g., prevalence of disease) and will persist indefinitely. Regime shift: future STEP alternates according to 30-year cycles in ocean climate. In poor climatic periods (simulation years 1997–2005, 2036–2065, and 2096), STEP selected from its posterior distribution. In good climatic periods (simulation years 2006–2035 and 2066–2095), STEP = 0.0.
Pre-removal period before drawdown	3 years 8 years
Juvenile survival rate after drawdown	Survival rate through drawdown reach = 0.89; average in-river survival rate (including unimpounded reach and four remaining dams and reservoirs) = 0.51 (CRiSP) and 0.39 (FLUSH). Survival rate through drawdown reach = 0.61; average in-river survival rate = 0.35 (CRiSP) and 0.30 (FLUSH).
Transition period after drawdown	2 years: assumed aquatic invertebrates re-establish quickly after major disturbances (e.g., Gore 1979); predators exert little pressure on juvenile salmonids in free-flowing conditions (Ward et al. 1995). 10 years: estimated time required for re-establishment of benthic macroinvertebrate habitat and communities (Pacific Northwest Laboratory 1995) and for stabilization of feeding behaviour and population dynamics of predators.
Ocean and in-river harvest	1. Ocean: Sample from 1985–1996 harvest rates <sup>a</sup> In-river: Current schedule <sup>b</sup> 2. Ocean: Reduced by 15% relative to 1985–1996 In-river: Current schedule 3. Ocean: Reduced by 50% relative to 1985–1993 In-river: Current schedule 4. Ocean: Reduced by 50% relative to 1985–1993 In-river: 50% reduction at lower Snake River run sizes, no increase in harvest rates until escapement exceeds 3750 spawners 5. Reduced by 75% relative to 1985–1993 In-river: Same as scenario 4
Upstream survival rates	Affected by construction and removal of dams Not affected by construction and removal of dams

**Note:** All assumptions except harvest and upstream survival rates were formally incorporated into the decision analysis; harvest and upstream survival assumptions were explored through sensitivity analyses. Alternative assumptions took the form of different parameter values in the passage or life-cycle models. CRiSP, Columbia River Salmon Passage; FLUSH, Fish Leaving Under Several Hypotheses.

<sup>a</sup>The U.S.–Canada Pacific Salmon Treaty was implemented in 1985.

<sup>b</sup>Based on the 1996–1998 Management Agreement for Upper Columbia River Fall Chinook.

### Decision analysis framework

The six uncertainties and their alternative hypotheses resulted in 16 possible outcomes (probabilities of exceeding survival and recovery thresholds) for action A2 (maximize transportation) and 128 possible outcomes for action A3 (natural river drawdown of four Snake River dams). These 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) analysis to the complete set of outcomes for all actions (Breiman et al. 1984; Watters and Deriso 2000). We used box-and-whisker plots of distributions of outcomes to compare the performance of actions across the range of uncertainties.

### Sensitivity analyses

We conducted sensitivity analyses to explore how results were affected by assumptions about future harvest management and effects of dams on adult survival rates during upstream migration (Table 3). Alternative harvest strategies incorporated reductions in ocean harvest rates on Snake River fall chinook ranging from 15 to 75% (relative to harvest rates occurring since 1985) in conjunction with both current in-river harvest rates and a hypothetical, conservation-based in-river harvest schedule.

Model runs for the drawdown action (A3) assumed that average upstream adult survival rates would increase from 0.42 to 0.83 following drawdown. We explored an alternative assumption in which upstream survival rates were insensitive to the presence of dams, have remained at current levels (averaging around 0.42) throughout the historical period, and will remain at current levels if drawdown occurs. This assumption required re-estimation of historical recruitment because the original estimates had assumed that upstream survival rates declined when the dams were completed in the early 1970s. Because this assumption changed recruit estimates, it affected results for both the maximize transportation (A2) and the drawdown (A3) action.

## Results

### Retrospective analysis

We selected the following model structure for prospective modeling:

$$(5) \quad R_t = (S_{wild_t} + E \cdot S_{other_t}) e^{\alpha} e^{-b(S_{wild_t} + E \cdot S_{other_t})} e^{-M_t} \\ \times (DP_{bt} + 1 - P_{bt}) e^{-STEP} e^{g \cdot e_t^{(des)}} e^{e_t'}$$

where the terms are defined as in eqs. 1 and 2. Selection of model components was based on the relative fits of the 30 alternative models to the spawner–recruit data (as measured by AIC and BIC scores; Table 2) and on the need to allow for implementation of alternative hypotheses about passage- and nonpassage-related factors.

The selected model in eq. 5 had a relatively complex structure that included passage model values for  $M$  and  $P_{bt}$ . These models had better fits than the simplest two-parameter models but the same or worse fits than models of intermediate complexity (compare models 1, 13, 25, 27, 29, and 30 in Table 2). However, including passage models provided convenient mechanisms for modeling direct effects of hydro-system actions. We included both CRiSP and FLUSH passage models as alternative hypotheses, although CRiSP estimates provided a better fit than did FLUSH (model 1 vs. model 13). We estimated  $E$  from spawner–recruit data to account for an interaction between  $E$  and  $D$  estimates, al-

though estimating  $E$  produced similar (using AIC) or worse (BIC) fits than fixing  $E$  at 0.7 or higher (model 1 vs. models 10–12; model 13 vs. models 22–24).  $E$  and  $D$  were related because hatchery production and the intensity of transportation increased around the same time in the historical period (Table 1). Therefore, a low  $E$  (ineffective hatchery spawners) required higher post-Bonneville survival of transported fish (higher  $D$ ) to produce observed spawner–recruit survival rates. Equation 5 also included a climate factor based on joint Snake River and Deschutes residuals. This approach resulted in a better fit than either ignoring climate effects or using an alternative approach (model 1 vs. models 2 and 5–8; model 13 vs. models 14 and 17–20).

There was an interaction between  $D$  and STEP. High mortality of transported fish in the estuary and ocean (implied by low  $D$  values obtained when  $D$  was estimated from historical spawner–recruit data) explained declining recruitment after transportation began in brood year 1975 and left little mortality to assign to STEP. Consequently, STEP = 0 whenever  $D$  was estimated. This suggested that the increase in mortality in the early 1980s (Fig. 2) coincided more closely with the increase in transportation that occurred at that time (Table 1) than with any STEP increase in mortality in 1976 or 1970. Conversely, when  $D$  was fixed at a relatively high value (e.g., models 9 and 21), the model estimated a non-zero value for STEP to account for declining recruitment after brood year 1975. This interaction between  $D$  and STEP led to two forms of eq. 5 for use in prospective modeling. In the Fall- $D$  model,  $D$  was estimated and STEP was set to zero. This form was most consistent with spawner–recruit data and with the hypothesis that the ineffectiveness of transportation best explained observed patterns in survival anomalies. In the Fall-S model,  $D$  was a fixed input value and STEP was estimated starting in brood year 1976 (this start year was more consistent with the data than 1970; model 27 vs. model 28). Specifying a fixed input value of  $D$  acknowledged that future  $D$  values may differ from historical estimates and allowed the model to estimate a measurable STEP for representing the hydro, stock viability, and regime shift explanations for observed patterns in survival.

Finally, PATH explored the effects of individual years of spawner–recruit data. Data from brood year 1991 had a large influence on estimates of  $D$  (values were higher if 1991 data were omitted; Hinrichsen 2001). Survival rates were much lower in that year than in other years (Fig. 2), whereas the proportion transported and the proportion of hatchery fish spawning in the Snake River were both high (Table 1). We did not exclude 1991 data from the prospective analyses because there were no anomalous circumstances (e.g., higher measurement error than in other years) to suggest that it should be.

### Prospective analysis

#### Relative effects of uncertainties

The CART analysis showed that the assumed effects of the actions had little influence on probabilities of exceeding the survival threshold of 300 spawners over 24 years. This was largely because the survival improvements assumed to result from drawdown did not take effect until 7–20 years after the start of the simulations. Hypotheses about the effec-

tiveness of transportation ( $D$  values) and passage survival rates (passage models) accounted for virtually all of the relatively small variance in this metric. CART results were different for the longer-term performance measures (probability of exceeding survival and recovery thresholds of over 100 and 48 years, respectively). There, differences in assumptions associated with the transportation and drawdown actions (such as the assumed increase in juvenile and adult survival rates associated with drawdown) were the most important influence in determining model results.  $D$  hypotheses had the next largest effect on the long-term measures, followed by hypotheses about post-Bonneville mortality (STEP). The combined effects of the actions and these hypotheses accounted for virtually all of the variance in the 100-year survival and 48-year recovery metrics.

#### Performance of actions across uncertainties

Actions A2 (maximize transportation) and A3 (natural river drawdown of four Snake River dams) produced similar distributions of probabilities of exceeding the survival threshold of 300 spawners (Fig. 5). Means and medians of the distributions were generally slightly higher for A3 but greater than 0.9 for both actions. The high probabilities of exceeding the survival escapement threshold were not surprising given that the average abundance of natural spawners from 1993 to 1997 (411 fish; Table 1) exceeded the threshold.

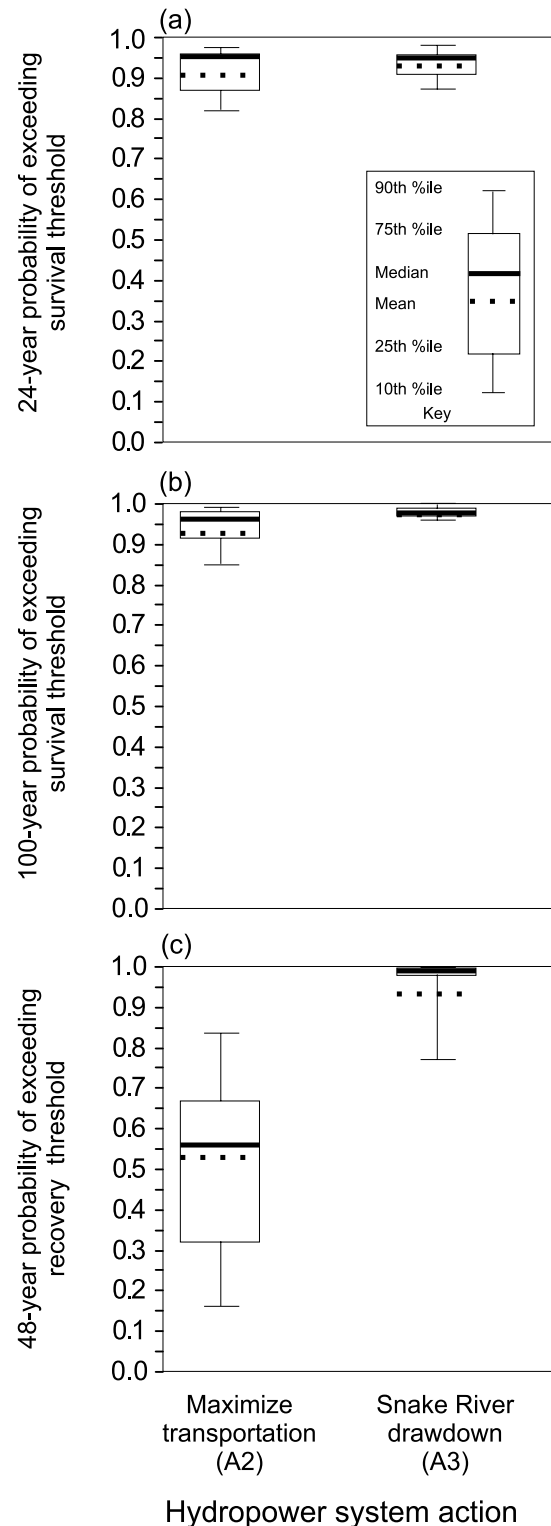
Exceeding the recovery threshold of 2500 spawners proved to be much more difficult to achieve. This threshold was similar to the average escapement over the complete time series, which included some years with high escapements (Table 1). The distribution of probabilities of exceeding this threshold over 48 years for A2 (maximize transportation) was broad with a mean of 0.53 and a median of 0.56. With A3 (drawdown), however, the distribution was highly skewed and around 80% of the 128 combinations of assumptions projected that the recovery threshold would be exceeded with a probability of 1. The mean and median of the distribution were both close to 1 with action A3.

The overall ranking of the transportation (A2) and drawdown (A3) actions, based on probabilities of exceeding recovery thresholds, was unaffected by the  $D$  hypotheses (Table 4). Probabilities for A3 were higher than those of A2 with all  $D$  hypotheses, although the difference was smaller when  $D$  was assumed to either improve over time (as in hypothesis D1, which assumed a threefold improvement in prospective  $D$  values over the retrospective value) or remain high and constant (as in hypothesis D2, where  $D = 1.0$ ). Further sensitivity analysis showed that for A2 and A3 to produce similar outcomes,  $D$  must either improve by 3.5-fold over retrospective values or remain constant at 1.8 or higher.

#### Sensitivity analyses

In general, alternative assumptions about harvest and upstream survival rates did not affect the overall ranking of actions (Table 4). With hypotheses D1 and D2 and the most extreme reductions in ocean and in-river harvest rates (scenarios 4 and 5), the increases in probabilities for A2 were large enough to approach base results (i.e., with no change in harvest rates) for the drawdown action A3. However, with hypotheses D3 and D4, base A3 results still exceeded those of A2, regardless of the assumed reductions in harvest rates.

**Fig. 5.** Distributions of probabilities of exceeding (a) the survival threshold (300 spawners) over 24 years, (b) the survival threshold over 100 years, and (c) the recovery threshold (2500 spawners) over 48 years for the two hydrosystem actions. There were 16 outcomes for the transportation action and 128 for the drawdown action. Distributions of 24-year and 100-year survival metrics were similar for the two actions, but drawdown produced higher and less variable probabilities of exceeding the recovery escapement threshold than did maximizing transportation.



**Table 4.** Probabilities of exceeding the recovery threshold over 48 years under different *D*, harvest, and upstream survival assumptions.

<i>D</i> hypothesis	Base assumptions <sup>a</sup>		Harvest scenarios (A2 results)				Alternative upstream survival rate under drawdown <sup>b</sup>	
	A2	A3	2	3	4	5	A2	A3
D1	0.87	1.00	0.89	0.95	0.97	0.97	0.92	1.00
D2	0.68	0.83	0.70	0.76	0.81	0.83	0.42	0.39
D3	0.28	1.00	0.31	0.38	0.44	0.47	0.17	1.00
D4	0.34	1.00	0.37	0.48	0.56	0.61	0.22	0.91

**Note:** Harvest scenarios and *D* hypotheses are defined in Table 3. Results for all *D* hypotheses are averaged over passage model and (for drawdown only) pre-removal, juvenile survival rate, and transition period hypotheses. Results for D2 and D4 are also averaged over the three post-Bonneville mortality (STEP) hypotheses. Only results for the maximize transportation action (A2) are shown for the harvest scenarios because results for the drawdown action (A3) under current harvest conditions were already high, leaving little room for improvement by reducing harvest rates. Results for action A3 were higher than A2 under most sets of assumptions.

<sup>a</sup>Results in these columns assume harvest scenario 1 from Table 3 and that upstream survival rates declined when Snake River dams were completed in the early 1970s and will improve if the drawdown action (A3) is implemented.

<sup>b</sup>The alternative assumption was that upstream survival rates were unaffected by dams, have remained at current levels throughout the entire historical period, and will remain at current levels if the drawdown action (A3) is implemented.

Assuming that upstream survival rates were unaffected by dams generally resulted in reduced probabilities of exceeding recovery thresholds for both the transportation (A2) and drawdown (A3) actions (Table 4). The effect of this alternative assumption on relative outcomes for the two hydrosystem actions depended on the *D* assumption. With D1 and D2, the alternative upstream survival rate assumption resulted in recovery probabilities for A2 that approached or exceeded those of A3. With low *D* values (D3 and D4), probabilities of exceeding the recovery threshold for A3 were still higher than those of A2.

## Discussion

The paucity of important spawner–recruit, juvenile survival, and transportation data resulted in a simpler life-cycle model, a much simpler decision tree, and a smaller number of possible outcomes of actions for fall chinook (16 for maximize transportation and 128 for drawdown, compared with 240 and 1920, respectively, for spring–summer chinook). We found that decision analysis was just as important in this data-poor situation as it was in the data-rich spring–summer chinook analysis because it helped us to focus analyses of limited data on testing specific hypotheses that underlie alternative decisions, rather than hypotheses that were ultimately less relevant to decision making. Moreover, the sensitivity analyses conducted as part of the decision analysis were useful for identifying key information gaps and prioritizing future research and potential adaptive management experiments.

The most significant consequence of the lack of information for fall chinook was that we were unable to draw firm conclusions about the long-term effectiveness of transportation. Although our results for the transportation action (as measured by the probability of exceeding the recovery threshold) were generally worse than results for drawdown, the difference in performance between the two actions was highly dependent on assumptions about the relative survival rate of transported fish once they are released from the truck or barge (i.e., the value of *D*), future harvest rates, and the effects of dams on upstream survival rates.

Decision makers may find these results unsatisfying because they do not provide unambiguous policy direction. However, they have at least two useful implications. First, they clarify what decision makers must assume about the relative likelihood of alternative *D* hypotheses, the improvement in upstream survival rates following drawdown, and the feasibility of implementing dramatic changes to ocean and in-river harvest schedules to justify a particular course of action. Second, the results demonstrate the relative biological risk associated with the two actions. Both the maximize transportation and drawdown actions produced high probabilities of exceeding the survival threshold of 300 spawners, although this was not a particularly difficult objective to meet. However, the drawdown action resulted in higher probabilities of exceeding the recovery threshold (2500 spawners) than did the transportation action over a wide range of uncertainties and assumptions. Therefore, if a hydrosystem decision must be made now, the most biologically risk-averse action would be to implement drawdown because it was projected to produce favorable outcomes over the widest range of uncertainties and assumptions.

The existence of uncertainty is often used to justify delaying a decision and maintaining the status quo because of the perceived risk of acting when a favorable outcome cannot be guaranteed (Peterman and Anderson 1999). Our results show that the maximize transportation action (which is most similar to status quo operation of the hydropower system) incurs significant risk of not achieving long-term recovery goals. However, if a decision must be delayed, we suggest that the interim focus should be on reducing uncertainty in *D* values using transportation studies. These may be more feasible in the near future as supplementation with hatchery-raised fish increases the number of fall chinook spawners returning to the Snake River. Future transport studies will not provide useful data on historical *D* values, but they could be used to narrow the plausible range of prospective *D* values and eliminate extreme hypotheses such as D3 (*D* = 0.05) and D2 (*D* = 1.0). Once a plausible range of prospective *D* values was identified, one could perform sensitivity analyses of prospective:retrospective *D* ratios to see what the retrospective *D* would have to have been for escapement under the transportation action to exceed the recovery threshold. Then,

those retrospective *D* values can be compared with available information such as spawner–recruit data to see if the implied historical effectiveness of transportation is consistent with historical patterns in overall survival rates.

## Acknowledgements

We thank PATH participants for their contributions to these analyses. In particular, we thank Howard Schaller, Olaf Langness, Phaedra Budy, and Eric Tinus for reconstructing spawner–recruit estimates from field data. Rich Zabel, Paul Wilson, Earl Weber, and Nick Bouwes provided passage model runs. We also thank the PATH Scientific Review Panel for their comments and suggestions throughout the PATH process and three anonymous reviewers for their helpful comments on an earlier draft of this paper. This work was funded by the 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.

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