

Judging Cost-Effectiveness of Management of Snake River Salmon: Response to Halsing and Moore

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Introduction

Halsing and Moore (2008) used Snake River spring/summer Chinook salmon (*Oncorhynchus tshawytscha*) as an example to present a synthesis of biological and economic information to develop a cost-effectiveness tool for assessing management alternatives for threatened or endangered species. Although we believe that elements of their approach could be useful to prioritize management alternatives and illuminate trade-offs between biological benefits and economic costs, we fear that their analysis may be of limited utility for Snake River anadromous salmonid management. Halsing and Moore used outdated, inferior models and parameter estimates to simulate the biological responses they used to rank cost-effectiveness of management alternatives, which depended on small differences between estimated population growth rates. They relied on a precision in estimated population growth rate unwarranted by the data and applied inconsistent economic analysis assumptions across scenarios, casting doubt on their cost-effectiveness findings.

Biological and Management Modeling

The passage model and range of differential transportation mortality (D) estimates Halsing and Moore (2008) used in the analysis may mislead decision makers about the relative biological benefits of smolt transportation versus in-river management options. They describe the passage model they used to estimate the effects of in-river measures on second-year survival rates Columbia River Salmon Passage (CRiSP) as the “most data-driven” model for this research. In fact, it has been criticized for being

too complex and overparameterized, with too many theoretical assumptions that lack empirical evidence, by 2 independent scientific review panels (Peters et al. 1998; ISAB [Independent Scientific Advisory Board] 2006). Neither National Oceanic and Atmospheric Administration (NOAA) Fisheries (Zabel et al. 2008) nor U.S. Fish and Wildlife Service (Schaller et al. 2007) uses mechanistic predator-based models of juvenile salmon migration such as CRiSP; instead, they prefer predictive models that relate empirical estimates of survival rate and migration velocity to environmental variables. Juvenile survival through the hydrosystem as modeled by CRiSP is notoriously insensitive to river flow and spill at dams, compared with less complex, more empirical models (Peters et al. 1998).

Newer, more rigorous estimates of differential transportation mortality (D) do not support the higher values of the range modeled by Halsing and Moore. Schaller et al. (2007), unlike Williams et al. (2005), account for interannual variation in sample size and estimate the mean of D over a 10-year period for wild Snake River spring/summer Chinook salmon (SRSSC) to range from 0.30 to 0.48, depending on point of transport. The mean D value for all fish, weighted by the proportions transported at each project, is 0.39. This suggests that little credibility should be given to the higher D values (0.7 and 1.0) modeled by Halsing and Moore. Overestimation of D leads to overestimation of the biological and cost-effectiveness of tern removal and underestimation of the benefits of strategies that minimize or discontinue transportation.

The degree to which and the mechanisms by which the hydrosystem causes latent mortality of juveniles in the estuary and early-ocean life stage (s_e) have been highlighted as critical uncertainties (Kareiva et al. 2000; Wilson 2003). Latent mortality may be explained, in part,

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Table 1. Cost-effective management alternatives and marginal analysis for Minam River index stock of Snake River spring/summer Chinook salmon ($D = 0.553$) with the use of 2 significant figures for λ .

Alternative number	λ	Cost	$\Delta\lambda$	Δcost	$\Delta\text{cost}/\Delta\lambda$
A3	0.86	-11.211	NA	NA	NA
C3	0.87	-10.986	0.01	0.225	22.5
D3	0.88	-6.681	0.01	4.305	430.5
A2	0.89	-2.25	0.01	4.431	443.1
C2	0.90	-2.025	0.01	0.225	22.5
D2	0.91	2.28	0.01	4.305	430.5
D10	0.92	172.259	0.01	169.979	16997.9
C17	0.93	208.394	0.01	36.135	3613.5
B17	0.94	212.473	0.01	4.079	407.9
A16	0.96	207.306	0.02	-5.167	-258.4
C16	0.97	207.531	0.01	0.225	22.5
B16	0.98	211.611	0.01	4.08	408.0

*The λ and cost values are from Haling and Moore (2008, Table 4).

by migration delays that cause in-river migrating juveniles from the Snake River to arrive at the estuary days or weeks later than they would in the absence of dams, with later-arriving migrants showing lower smolt-to-adult survival rates than earlier migrants (Muir et al. 2006; Waples et al. 2007). Haling and Moore modeled latent mortality reduction only for the breach alternative; however, several other management alternatives that would speed the juvenile migration (spill, flow augmentation, surface passage technology, and reservoir drawdown) also have the potential to reduce latent mortality.

Haling and Moore did not model actions contemplated in the recent Biological Opinion on the Federal Columbia River Power System. For instance, the Reasonable and Prudent Alternative recommendations include drawing down John Day, the largest lower Columbia reservoir, to minimum irrigation pool, with the intent of speeding juvenile migration through the lower Columbia River, providing surface spillway weirs at most dams to reduce passage delay and the portion of juvenile migrants going through the turbines, and adjusting transportation operations in season depending on date or flow volume (NOAA Fisheries 2008). Instead, Haling and Moore included implausible scenarios, such as breaching Snake River dams while transporting juvenile migrants collected at McNary Dam (MCN). Transportation of spring migrants at MCN was halted due to low transport to in-river ratios (NMFSS [National Marine Fisheries Service] 1995). It is unlikely that aggressive measures such as dam breaching would be combined with transporting fish that have made it to the lower Columbia River.

Haling and Moore omitted the consideration of freshwater habitat restoration, saying that biological models are not available to estimate changes in survival. Tributary habitat assessment tools have been developed with data from the Columbia River basin to evaluate the po-

tential for habitat projects to increase population growth rates. Budy and Schaller's (2007) analysis indicates that there is little room for improvement for roughly half of SRSSC indicator populations, including the Minam River, which is featured in Haling and Moore (2008). We are also concerned about Haling and Moore's parameterization of the projection matrix. Their Table 2 indicates that, unlike Kareiva et al. (2000) and Wilson (2003), they used an age-invariant (age 3) fecundity.

Estimating Cost-Effectiveness

To compare cost-effectiveness of the alternatives, Haling and Moore estimated population growth rate (λ) of SRSSC for a suite of actions and estimated of the cost of those actions relative to a baseline scenario. The estimates of λ are reported to 3 decimal places; the set of "cost-effective" alternatives in Tables 4 and 5 is highly dependent on estimating λ accurately to 3 significant figures. It is not clear how Haling and Moore selected the "growth-rate constraints" used in seeking cost-effective alternatives described in Eq. 4. For instance, in Table 4, A5 is not identified as a cost-effective alternative, perhaps because its λ differs from that of A3 by only 0.001. Conversely, D7 and D2 are both included in the cost-effective set, but their estimated λ s also differ by only 0.001 (according to Table 4). In addition, λ values reported in Tables 4 and 5 differ for alternatives C2, C3, C16, D2, D3, D7, D10, and D16. This is particularly troublesome, in part, because cost-effective transitions between alternatives in Table 5 are judged by the incremental change in cost and λ ($\Delta\text{cost}/\Delta\lambda$) from each alternative to its successor. Estimates of $\Delta\lambda$ often have a significant figure only in the thousandths place; minute errors could render estimates of $\Delta\text{cost}/\Delta\lambda$ wildly inaccurate.

Discrepancies between λ values in the 2 tables may represent only typographical errors, but the use of such small differences in λ to discriminate between the utility of alternatives is still inadvisable. Passage-model survival rates estimated under different hypothetical futures or estimates of survival increase due to reduction in predation by terns are unlikely to be accurate to 3 or more significant figures. Errors, both random and systematic, of the component parameter values of the matrix entries compound to render distinctions between scenarios judged by differences in λ in the thousandths place dubious.

Wilson (2003) reports λ only to 2 significant figures because of slight differences from Kareiva et al.'s (2000) values in some index stocks in the third significant figure, due to rounding error and differences in the algorithms used to solve the characteristic equation of the matrix. Fitting the projection matrix to estimates of generational (i.e., multiyear) productivity requires iteratively and simultaneously modifying λ and generation time (T)

(Wilson 2003, Eqs. 4 & 5). These equations can be fitted by a number of procedures; the procedure and tolerance chosen can influence the exact values of λ and T resulting from a particular set of matrix entries. Considering the cumulative effect of estimation and fitting error, it is imprudent to sort actions by cost-effectiveness on the basis of differences in λ in the thousandths place. The use of 2 significant figures instead of 3 to sort cost-effective alternatives to recreate Haling and Moore's Table 5 results in a different set of cost-effective alternatives and ranking of effectiveness of transitions between adjacent alternatives (Table 1).

The set of cost-effective alternatives in Table 1 differs from Table 5 of Haling and Moore in that B2, B3, D7, and D16 drop out, whereas B17 and C17 are included. The most clearly cost-effective transition is from B17 to A16, where λ increases, whereas cost actually decreases. Conversely, Haling and Moore's Table 5 indicates 6 transitions, with positive, similarly low values of $\Delta\text{cost}/\Delta\lambda$ as the most cost-effective.

In their analysis of cost-effectiveness, Haling and Moore did not consider the impacts of the actions analyzed on other species subject to regulatory authority (other salmon species, Caspian terns [*Hydroprogne caspia*]). Ignoring the impacts of the alternatives on the 3 other listed species of Snake River anadromous salmon (fall Chinook salmon, steelhead, and sockeye salmon) limits the utility of Haling and Moore's single-species cost-effectiveness analysis. Some actions considered, such as dam breaching, would benefit all species, whereas other actions, such as harvest reductions or flow augmentation, could not be configured to provide substantive benefit to all species. In addition, Haling and Moore note that dam breaching would provide river-based recreation benefits, but despite referencing Loomis's (2002) analysis that the recreation benefits alone could offset the total costs of dam breaching, no benefit is credited in the net dam-breaching costs (e.g., Table 5). Nevertheless, the cost used for the dam-breaching alternative includes foregone reservoir-based recreation (p. 344).

Conclusion

Haling and Moore's cost-effectiveness analysis relied on distinctions between the biological consequences of alternative actions that are too fine, given the modeling tools and data available, and likely inaccurate in their ranking of relative merit due to the use of inferior models and parameter estimates. They modeled implausible management scenarios and omitted others more likely and used inconsistent accounting of costs and benefits between alternatives. Despite the authors' claim that their analysis takes recovery as a legal requirement, they included alternatives that they estimated would decrease popula-

tion growth rate. Although elements of their approach may be useful to evaluate cost effectiveness, the published analysis should not be used to guide Snake River salmon recovery effort.

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