

Submission 21

Extra Mortality Hypotheses

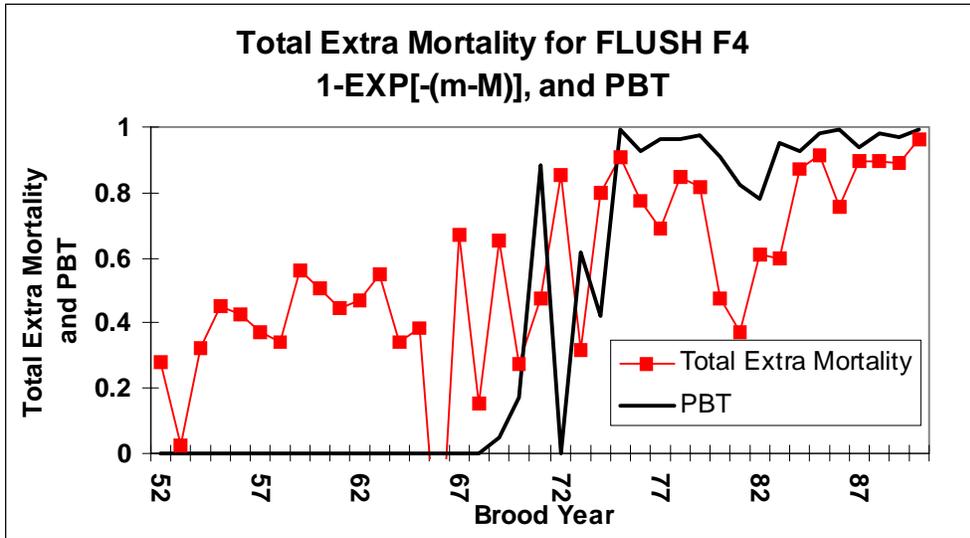
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I. Extra Mortality Hydrosystem Hypothesis- λ_n

Pertains to Appendix 3 and Hinrichsen 7/28/98 Comments.

Hinrichsen and Paulsen 1998 imply that the lack of a relationship between $1-\lambda_n$ and $1-V_n$ in the 1975-1990 brood years nullifies the hydrosystem hypothesis. First, $1-\lambda_n$ (extra mortality of in-river fish) is only a small component of the hydrosystem hypothesis. The direct mortality and the delayed mortality of transported fish comprise a larger component of the hydrosystem hypothesis. The argument that $1-\lambda_n$ is not correlated with $1-V_n$ in the 1975-1990 brood years is confounded by the extremely high proportion of fish that survive to Bonneville dam that are barged (*See below*, Figure 1). λ_n is defined as $\exp(-m) / \omega$, which means it must explain what is left over, after m and ω have been estimated. The extra mortality of in-river fish is the remaining extra mortality term after the total mortality term has been reduced for direct mortality (estimated by the passage models) and transport extra mortality (passage model and transport model estimate). Therefore, $1-\lambda_n$ is the residual of mortality after MLE total mortality estimate has been reduced by passage model estimates. Given the magnitude of the fish transported in the brood years 1975-1990, $1-\lambda_n$ would be heavily influenced by the assumptions for the transport extra mortality assumptions in the passage models. In fact, after the onset of transportation, the structure of the delta and alpha models is arbitrary with respect to how one defines system survival and D . One could have expressed system survival in transport fish equivalents, the order of the λ s in the ratio D would have been reversed, and λ_t would be left to pick up the slop from estimation problems of the other parameters. The lack of a relation of $1-\lambda_n$ with $1-V_n$ is not evidence that a hydrosystem-related extra mortality does not exist, but the problem may be that $1-\lambda_n$ is a residual of all the other passage model mortality and transport effectiveness assumptions.



° Figure 1. Total extra mortality (1-EXP[-(m-M)]) for FLUSH F4 model run compared to proportion of smolts below Bonneville Dam that were transported (PBT), brood years 1952-1990.

Given the structure of the models, and the fact that the hydro hypothesis is about prospective survival relative to that of retrospective survival in a migration year with similar natural runoff estimates (i.e. similar amount and timing of snow melt), it is impossible to test with retrospective data. We can consider the relationship between λn and $V(n)$ retrospectively, but this suffers from the limitations noted above. Even if it could be legitimately demonstrated that $1 - \lambda n$ shows no clear relationship to $1 - V(n)$ with existing data, it would not demonstrate that post-Bonneville mortality of non-transported fish is not related to the conditions they experience in the hydrosystem. As Williams et al. (1998) hypothesize in the Appendices to the WOE report, “[T]he large number of hatchery fish [in] the Snake River Basin likely increases the stress levels on migrant chinook in areas where fish congregate, such as in the forebays of dams before fish pass (particularly at Lower Granite Dam), in collection raceways, and in transportation barges.” If these mechanisms are important, under status quo scenarios (A1 and A2), delayed mortality would change little from past values, since wild spring chinook would still experience being confined in small areas and tight spaces with hatchery chinook and steelhead. There would be some small improvement over the retrospective (water years 1977-92) because of increased spills at many projects. However, removal of four dams (scenario A3) is expected to result in a large decrease in delayed mortality, because of the elimination of the crowding effects of four dams.

The hydro-related extra mortality hypothesis was implemented in a manner such that the decrease in extra mortality as $V(n)$ increased was relatively small. The lack of sensitivity is shown in ESSA (July 29, 1998), “Sensitivity to weightings on key alternative hypotheses”. Implementation of the hypothesis with FLUSH results in much greater benefit for A1 and A2 than A3 (Figure 14 in ESSA 1998). In summary, the lack of evidence of a positive relation between passage model Vn and λn (residual survival) does not negate the hydro extra mortality hypothesis, but may warrant alternative formulations.

The John Day Middle Fork salmon sensitivities summarized in Hinrichsen 7/28/98 figure 2 again focus on changes in λn . This sensitivity of λn 's also is burdened by all the problems we listed above for Vn vs λn .

II. Regime Shift Extra Mortality Hypothesis

The draft Weight of Evidence (WOE) Report provided a comprehensive summary of evidence for and against the regime shift hypothesis presented in PATH to date. In addition to the draft WOE report and comments previously submitted (Petrosky et al. 1998, Appendix 15), we provide the following comments.

First, we reiterate that the “regime shift hypothesis is really a selective regime shift hypothesis (see Schaller et al. comments in preliminary decision analysis report, Marmorek and Peters 1998, p. A107). As framed, it does not address why the climate regime shift selectively reduces survival rates more for Snake River stocks than for downriver stocks, after 1975, unrelated to hydrosystem passage. Support for the selective regime shift hypothesis requires evidence that Snake River spring/summer chinook stocks are particularly vulnerable to climate variability, unrelated to their unique hydrosystem experience.

The evidence presented to date to support a regime shift hypothesis has relied heavily on analysis of changes in salmon catch or escapement. Hinrichsen (1998; Appendix 17, Table 1) compares changes in catch for four Alaska stock groups (Western Alaska sockeye, Central Alaska sockeye, Central Alaska pink, and Southeast Alaska pink) to changes in run-size of Columbia River upriver spring chinook. We have previously commented on this data set (see Schaller et al. comments in preliminary decision analysis report, Marmorek and Peters 1998, p. A107-A110):

This summary table appears to be evidence that different species (sockeye and pink) from broad geographic areas (most of Alaska) responded similarly (using mean catch statistics) to climate regime shifts (major PDO polarity). It could be used to support the common year effect of the delta model. Possible mechanisms, rationale and evidence are needed to explain why, under this hypothesis, that very similar, adjacent stock groupings of stream-type chinook would have vastly different responses to climate regimes, unrelated to their unique hydrosystem experience.

The [Columbia River] data are for the aggregate wild and hatchery run-size of upriver spring chinook. No attempt was made here to sort out the effects of changing mainstem harvest rates (range 0.03 to 0.86) or of proportion wild fish in the run. If this [data set] is evidence of a regime shift on upriver aggregate spring chinook (including both Snake River and lower river stocks), it does not provide evidence in support of a selective regime shift hypothesis.

Note also that the stock performance measures being referenced in this rationale are indices of abundance (catch or run-size). Productivity and survival rate data would provide a stronger basis in support of a hypothesis, since abundance is a function of escapement and exploitation policies, as well as productivity and survival. There is recent evidence that changes in abundance indicators may be explained by changes in escapement and harvest management policies (Farley and Murphy 1997), rather than climate patterns.

In an analysis of catch data for 9 major sockeye salmon stocks from Alaska and northern British Columbia, Farley and Murphy (1997) found that a long-term shift in catch levels appeared to be the result of changes in escapement policy rather than an abrupt change in production dynamics of the North Pacific. Farley and Murphy (1997) also note that catch data analyzed at the stock level provides a more meaningful interpretation of catch patterns than at more aggregate stock levels because of differences in production dynamics between salmon stocks and because effective management requires decisions based on the individual stock level

Residuals from Spawner and Recruit functions are a more appropriate index of survival rates than using catch statistics. This concept was supported by R. Peterman’s comments at the July 30-31, 1998 PATH Weight of Evidence meeting.

Climatic influences on sockeye salmon productivity and survival rates appear to operate on fairly large regional scales. Adkison et al. (1996) compared alternative models of productivity (R/S) for sockeye stocks from Bristol Bay, Alaska and Fraser River, British Columbia (BC). In Bristol Bay, productivity co-varied among stocks, and appeared to be strongly related to fluctuations in climate. None of the models explained much of the variability in productivity for Fraser River stocks.

Peterman et al. (1997) found evidence that large-scale environmental processes can strongly influence survival rates of sockeye salmon on regional scales. Survival rate indices were calculated as the residuals from Ricker stock-recruitment models, and correlated among and between BC and Bristol Bay stocks. There were large positive correlations in survival rate indices among 9 stocks from the Bristol Bay region, and to a lesser extent among BC stocks, with a tendency toward negative correlations between Bristol Bay and BC stocks. This suggests that within each region, the interannual variability in survival rate indices is influenced by common environmental processes, but that these processes are distinct for the stocks of each region. The Bristol Bay stocks were genetically distinct (R. Peterman comments at WOE meeting), contradicting elements of the NMFS hypothesis which states that fish from the two genetic stock groupings may have systematic differences in survival that change independent of each other under varying environmental conditions.

Year of ocean entry also appeared to be important in determining variability in survival rate of sockeye from the Bristol Bay region (Peterman et al. 1997). Bristol Bay sockeye smolts were a mix of age 2 (sub-2) and age 3 (sub-3). Survival rate indices of sub-2 adults were more highly correlated with those of sub-3 adults that had the same year of ocean entry (OEY) versus adults that had the same brood year. This indicates that the late freshwater or early marine life stage (shared by age groups with the same OEY) may be more important than the early freshwater life stage (shared by groups of the same brood year) for determining variability in survival rates.

L. Botsford, in the PATH, Preliminary Decision Analysis Report on Spring/Summer Chinook, Draft for Scientific Review Panel, February 1998, 'Comments on Regime Shift hypothesis':

It is clear that there was a change in the atmosphere in the north east Pacific in the mid 1970s, and considerable evidence that that change had a positive influence on biological productivity in the Gulf of Alaska. There are also mechanistic reasons to believe that there is an inverse relationship between the flows in the Alaska Coastal Current and the California Current. However, whether the change in the 1970s had an effect on salmon stocks in Washington, Oregon and California is just a hypothesis, and evidence regarding that hypothesis must come from the area where these fish are found. The Columbia River stocks spend their early life in the California Current, south of the bifurcation of the West Wind drift, so if that period is important we need evidence from salmon stocks in the California Current.

One piece of evidence on the effect of the regime shift on CCS stocks is the decline of Oregon hatchery survivals in the 1970s described Percy (1997). However, that same decline is not seen in the Coronado-Hernandez thesis (Fig. 11). Another piece of evidence (though marginal because of the location) is the decline in survivals of chinook stocks in the Georgia Strait (Beamish 199x). There is also a marginal correlation between zooplankton densities off southern California (where there are no salmon), and zooplankton densities in the Gulf of Alaska (Brodeur, et al. 1996?).

Some of the relationships presented in the current document are not rigorous evidence. In the paper by Mantua, et al. (1997) no "correlations" are computed between the PDO and Columbia

River stocks nor Washington-Oregon-California coho. There are also no correlations between a variable representing the regime shift and California Current salmon in Francis and Sibley (1991) nor in the Anderson (1996) review. As far as I know, none of the referenced papers contain statistical evidence relating California Current salmon stocks to the regime shift in the Gulf of Alaska.

Residuals from the preliminary fall chinook BSM show no evidence of a regime shift for the Hanford or Lewis stocks (See below and Deriso, personal communication).

If the regime shift were a plausible hypothesis to explain extra mortality, we might expect to see a change in the pattern of residuals from the stock recruitment function for the Hanford (HYURB) and Lewis River Wild (LRW) fall chinook stocks starting in (or around) brood year 1976 (ocean year 1977). However, no such pattern exists (see Figure 2 below). The reason these stocks are important is that the Hanford stock is only above four dams and the Lewis River stock is below all mainstem dams.

Further, the period effect was insignificant for the ANCOVA's of $\ln(R/S)$ as a function of spawners and period. Periods were defined as: \leq Brood year 1975, >1975 (tests for homogeneity of slopes were insignificant).

ANCOVA results				
	DF	MSE	F	P>F
Hanford				
Spawners	1	7.16	15.32	0.0006
Period	1	0.29	0.62	0.439
Lewis				
Spawners	1	12.13	33.23	0.0001
Period	1	0.57	1.57	0.2212

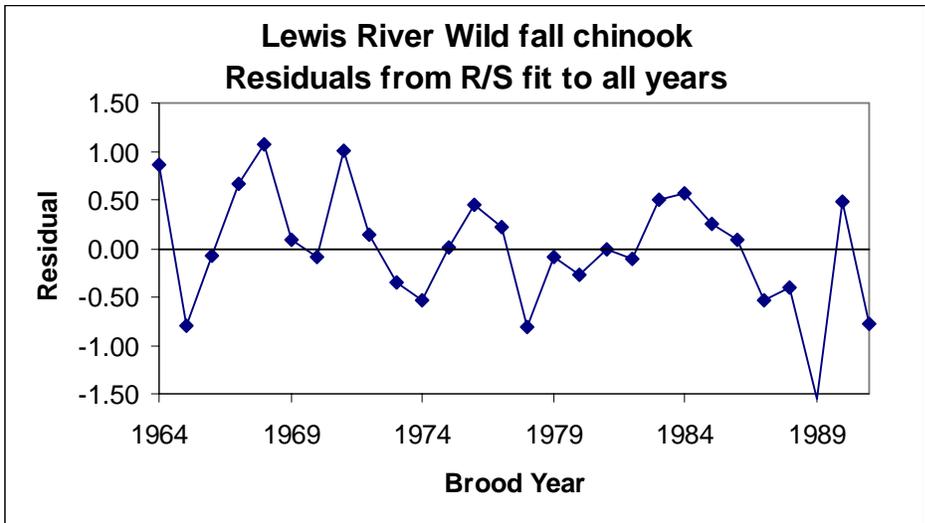
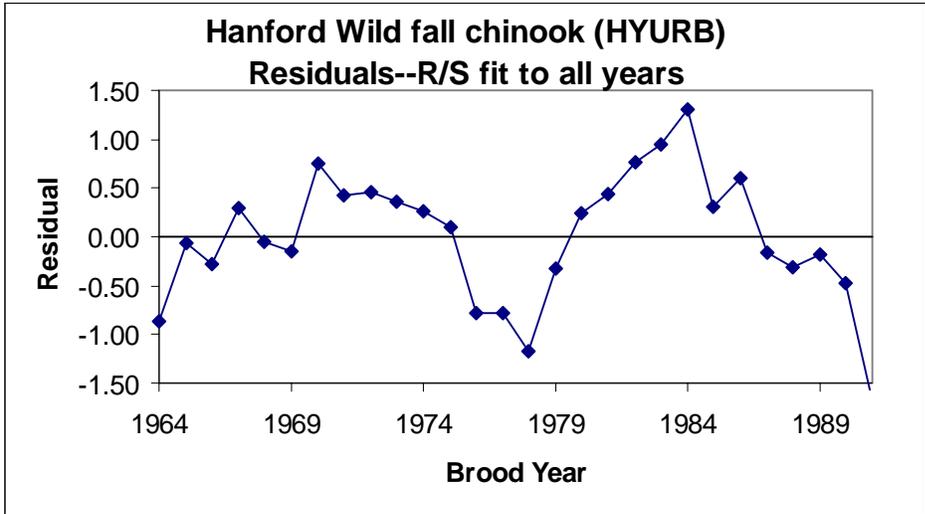


Figure 2. Residuals from $\ln(R/S)$ versus S for all years (BY 1964-BY 1991) from fall chinook retrospective analysis with the Hanford wild (HYURB) stock above (from Schaller et al. 1998) and the North Fork Lewis wild stock below (Langness et al. 1998) demonstrating lack of evidence for a regime shift. The regime shift is hypothesized to have begun in ocean year 1997 which corresponds to Brood Year 1976.

Problems with the PAPA index and the history of the environmental variable:

PAPA drift “index” is not a physical measurement, but actually a model prediction. As Anderson and Hinrichsen (1997) and Anderson (Submission 11, Appendices) point out, it has been computed for a hundred year record using the OSCURS ocean circulation model. The assumptions and mechanisms of the model have not been reviewed within PATH. There have been several incarnations of the environmental index included in the regime hypothesis. For a proposed flow/ocean environmental hypothesis, Anderson and Hinrichsen (1997a) suggest an equation for inclusion of the PAPA drift output in the alpha model that includes the OSCURS output (E) in two terms: a first degree E divided by river flow term, and a second degree E term (i.e. E² divided by river flow). In addition, they include an inverse flow term and a regional “drought index” term. Presently, the hypothesis includes a single term with E and a single (inverse) term with F (river flow at estuary), as well as terms representing avian predation and hatchery fish in the estuary. Previous incarnations used the North Pacific Index and ocean upwelling index (Anderson and Hinrichsen 1997b)

Productivity and Survival Rate Indices

Historically, Snake River stocks were the largest component of the aggregate upriver spring chinook run, followed by upper Columbia River stocks (Fulton 1968). Therefore, the negative effects of a regime shift hypothesized for before 1947 should be evident in survival rates of the aggregate upriver spring chinook run. The following comments are summarized from Schaller et al. 1996, PATH Retrospective, Chapter 3 addresses :

Spawner and recruit data of the aggregate upriver run of wild spring chinook for brood years 1939-1990 provided little or no evidence of a long-term, gradual decline in productivity and survival rate. Rather, the analyses provided support for hypothesis 3b, that the productivity and survival rate of upriver spring chinook remained fairly stable from early hydropower development (1939) until the era of major hydropower development (about 1970), when major declines began.

Analysis of covariance and least square means tests found no differences in productivity estimates between the periods 1939-1949, 1950-1959 and 1960-1969. Productivity estimates from the periods 1970-1979 and 1980-1990 were significantly less than any of the early periods. This aggregate provides a longer time series of R/S data than any of the index stocks. The indices of climate change over the Pacific Ocean, which Beamish et al. (1997) linked to sockeye salmon (*O. nerka*) production, varied widely from 1939 to 1970. Interestingly, the productivity of the aggregate remained fairly stable (relative to post-1970) through these decades and then decreased coincident with the period of major hydropower development and operation. Plots of survival rate indices for the aggregate upriver run also indicated the major declines in survival rate began about 1970.

Our conclusion that hydropower was a primary cause of productivity and survival rate declines of upriver stocks was conditioned on evidence that the estuary and early ocean conditions did not have a systematically different effect on survival for stream-type chinook stocks across regions of the interior Columbia River basin. This is reasonable in view of the similarity of these stocks, the overlap in time and space of these stocks during their early ocean residence (and beyond), and the broad-scale nature of climatic influences described in the literature.

There are several lines of evidence suggesting that the interior Columbia Basin stocks are exposed to similar estuary and ocean conditions, particularly during the critical first year. Beamish and Bouillon (1993) and others provided evidence that indices of climate over the north Pacific Ocean may play an important role in production of different species of salmon originating over a wide geographic range. Deriso et al. (1996) found evidence of a common year effect for index stocks of stream-type chinook from

the Snake River and lower Columbia River regions. Of the lower Columbia River stocks in this analysis, at least the John Day River and Warm Springs River spring chinook smolt timing appears very similar to that of Snake River spring and summer chinook. Smolts of these lower Columbia River, Snake River and upper Columbia River stocks migrate through the mainstem to the estuary primarily in late April and May (Lindsay et al. 1986, 1989; Raymond 1979; Hymer et al. 1992; Mains and Smith 1964). Current hypotheses regarding ocean survival of Pacific salmon generally focus on the juveniles' critical first months at sea (Pearcy 1988, 1992; Lichatowich 1993), where juveniles of these index stocks are most likely to overlap in time and space. Year class strength for these spring and summer chinook is apparently established, for the most part, within the first year in the ocean, as evidenced by the ability of fishery managers to predict subsequent adult escapements from jack counts (e.g., Fryer and Schwartzberg 1993).

Since it appears that Columbia Basin stream-type chinook share a common estuary and nearshore ocean environment and a more common ocean distribution than stocks evaluated by Beamish and Bouillon (1993), it seems highly unlikely that differential estuary and ocean conditions could explain systematic differences in stock survival. The ocean recoveries of coded wire tagged (CWT) stream-type chinook were infrequent (Berkson 1991). The few recaptures (62 recoveries from 8 release years) from hatchery stocks in both the Snake River (21 recoveries) and lower Columbia River (41 recoveries) were widely scattered from California to Alaska ocean fisheries (Pacific States Marine Fisheries Commission unpublished data). The average annual proportion of CWT recoveries from ocean fisheries north and south of the Columbia River mouth appears to be similar between the Snake and lower Columbia hatcheries.

Development of the Canadian storage projects in the upper Columbia River in the mid-1970s, and hydrosystem regulation have reduced flows during the spring smolt migration for all stream-type chinook (Raymond 1988). Since Columbia Basin stream-type chinook share a common lower river migratory corridor and estuary, changes that may have occurred due to the development of storage projects in the mid-1970s are unlikely to account for the differential decline in productivity and survival rates between the upriver and downriver index stocks.

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