

Last December, the National Marine Fisheries Service released a draft paper, known familiarly as the “All-H Study,” that examined the four Snake River population groups (or ESUs) of salmon and steelhead, with particular emphasis on how, together and separately, the so-called four H’s of habitat, hydropower, hatcheries and harvest, affected these stocks. At that time, the agency promised a more thorough analysis of the remaining eight Columbia Basin stocks. The result of that further analysis is this draft study, summarized below. Its is straightforward: all the Columbia River Basin salmon stocks are in a state of perilous decline, especially Upper Columbia Spring Chinook and Steelhead throughout its range. Put in starker terms: without substantial intervention, there is a greater than 50:50 chance that most of these ESUs will be extinct by the next century, some much sooner.

Thus, this draft report is a crucial management tool, quantifying for the first time the rates of declines for the Columbia River Basin salmonid stocks and articulating the management options available to arrest and reverse those declines.

A Standardized Quantitative Analysis of the Risks Faced by Salmonids in the Columbia River Basin

Since 1991, twelve salmonid Evolutionary Significant Units (ESUs) in the Columbia River Basin have been listed as threatened or endangered under the Endangered Species Act. Because these dozen ESUs share overlapping ranges, common waterways and common threats, it is neither practical nor wise to treat each ESU as a separate management problem. Instead, we need to broadly examine the entire Columbia River Basin and its salmonids. Until now, we have lacked reports presenting even the most basic population size and trend data for Columbia River Basin salmonids in a common format. Hence, standardized data synthesis and analysis is a key feature of this document. In addition, we present a standardized assessment of extinction risks and the magnitude of improvements required to mitigate these risks. Finally, to varying degrees, depending on what data is available, we begin to evaluate the potential effects of management actions aimed at different life stages or sources of mortality. The lessons learned from this synthesis of data and analyses are broadly sketched for the entire region.

A. Standard descriptions of data and summary metrics

Standardization requires the adoption of a consistent methodology. A lack of uniformity in the time periods for reporting or analyzing data, as well as methods of analysis, has made it impossible to compare different ESUs or different populations within the same ESU. Too often discussions regarding salmon become muddled because the parties involved discuss varying time intervals, or report data in different ways. In our approach to quantifying the risks faced by Columbia River Basin salmonids, we have chosen the following standards.

Time period: All analyses in this report use spawner counts or other population data from 1980 until current (or as current as is available). We have selected 1980 as a starting point because prior to that year the hydropower operations in the Columbia River Basin were not firmly

established. Secondly, the farther back in time one goes, the less likely the data are to represent current biological and environmental conditions, and it is the current situation that needs to be clearly described and addressed. One factor not included in these analyses is the influence of different scenarios for altered ocean conditions relative to the baseline time period of 1980 to current. The possibility that ocean conditions may improve relative to our current period, or deteriorate even further, may need to be considered when examining the policy options for certain ESUs.

Population counts and running sums: At annual intervals we report the raw number of spawners counted, taking care to document where these counts were made, and the running sum of spawners tabulated in a way that measures the total spawner population, including the “unseen fish” in the ocean as well as those counted on the spawning ground. In addition to aggregated counts for eleven ESUs (sockeye salmon are not examined), we report and analyze time series of counts for 57 different index stocks.

Annual rate of population change: For both ESUs and individual index stocks we estimate average annual rate of population change or “lambda.” Lambda, which incorporates year-to-year variability, is the best summary statistic of how rapidly a population is growing or shrinking. A lambda less than 1.0 means the population is declining; a lambda greater than 1.0 means the population is increasing.

Risk of extinction: By combining lambda with estimates of environmental variability it is possible to calculate “extinction risk metrics.” All extinction metrics are calculated on a 24- and 100-year timeframe. For index stocks, where our data represent entire population counts, we estimate extinction risks in terms of the probability of an adult population falling to only one spawner. For ESUs we calculate extinction metrics as the probability of a 90% decline after 24 years and after 100 years, because it is unlikely that entire ESUs have been accurately counted.

Key diagnostics: Our estimates of annual rates of decline and extinction risks depend on several assumptions about the structure of these data. The most important assumptions concern whether or not there is evidence of density-dependence in the time series and whether there are trends in the temporal pattern of recruits per spawner. We test for density-dependence in a formal statistical manner, but rely on simple graphs to portray trends in recruits per spawner. We are developing methods for estimating the significance of trends. If graphs indicate striking declining trends in recruits per spawner, then the standardized extinction-risk metrics will be underestimates, assuming that the trends continue into the future.

B. RESULTS OF ANALYSES

Overview of Annual Rates of Population Change and Risks of extinction

At the ESU level, the annual rates of population change were less than 1.0 for nine of the eleven ESUs and less than 0.9 for Upper Columbia Spring Chinook salmon, Middle Columbia steelhead, Upper Columbia steelhead, and Upper Willamette steelhead. The four ESUs showing lambdas less than 0.9 are decreasing at such a rapid rate, at least 10% per year, that we can

expect to see only tiny fractions of their already depressed populations surviving out to 24 years. It is worth noting that at the ESU level the salmonids showing the most dire risks of perilous declines were *not* Snake River Spring/Summer Chinook.

At the index stock level, the range of risks is broader, but the same overall picture is evident. For example, over two-thirds of the stocks exhibited lambdas less than 1.0, and one third had lambdas less than 0.9, with the average lambda across 57 index stocks at 0.95. Similarly, the extinction risks, in this case calculated as the probability of dropping to only one fish within 24 or 100 years, were also high, averaging 12% at 24 years and 58% at 100 years.

Overview of Improvements Needed to Mitigate Risks

Generally, ESUs or stocks with the most rapid rates of decline – the lowest lambdas -- require the most improvement to mitigate extinction risks. However, this generality is complicated by the fact that low populations and high environmental variability can exacerbate extinction risks beyond what might be expected from lambda alone. The magnitude of improvements required in lambda ranged from less than 1% to as much as 65%, with most values falling between 5% and 20%.

The more difficult task is exploring opportunities for improving lambda, *i.e.*, increasing the number of recruits per spawner. The well-known “four H’s” (hydropower, habitat, hatcheries and harvest) represent the human-influenced arenas in which management can be altered in hopes of recovering ESUs. But because these four H’s vary enormously in the areas occupied by different ESUs, it is unlikely that a simple prescription can be drawn up that fits all ESUs. For example, the number of dams per kilometer varies from 0.4 to 2.8 depending on the region associated with each ESU. Land use characterization also varies widely across regions occupied by ESUs, with some regions characterized by a high percentage of rangeland (Upper Columbia and Snake Rivers), urbanization (lower Columbia, and upper Willamette Rivers), or cropland (upper Willamette River). At the finer scale of index stocks, preliminary analyses indicate that three habitat variables at the subwatershed scale explain 60% of the variation in recruits per spawner: (1) percent of land classified as urban, (2) proportion of stream length failing to meet EPA water-quality standards, and (3) the ability of streams to recover from sediment flow events. Lastly, although nearly 100 hatchery facilities in the Columbia Basin release approximately 150 million smolts annually, the magnitude of this hatchery production varies by an order of magnitude among ESUs. The impact of this hatchery production is difficult to analyze because of the lack of large-scale controlled experiments. Some preliminary analyses suggests that in “poor ocean years” hatchery fish compete with wild fish and lower the survival rates of the wild fish.

In summary, there are no clear-cut analyses that allow confident predictions about likely improvements in lambda if actions are taken in hydropower, habitat or hatcheries. Generating such predictions is clearly a research priority. But right now, science cannot provide hard numbers on questions about how any ESU will respond to any particular management option, although the direction of effects and what would qualitatively represent an “improvement” are usually known. The one exception, of course, is harvest, because harvest is essentially a scheduled mortality, and analyzing changes in that mortality is straightforward. Harvest

reductions, some of which already have been initiated, are clearly capable of achieving large increases in lambda (20% to 30%) for Lower Columbia Chinook, Upper Willamette Chinook, and Snake River Fall Chinook ESUs. For the remaining ESUs in the Columbia River Basin the opportunities for improvements due to harvest reduction are much less substantial. It is important to make clear that this analysis most emphatically does *not* mean that harvest reductions are the thing to do, but only that it is easiest to predict the consequences of a harvest reduction as compared to actions in habitat, hydropower, or hatchery operations.

C. THE NEXT SCIENTIFIC STEPS

Critical Sampling Uncertainties

One of the greatest uncertainties does not involve the biology of salmonids; it is a simple counting problem. Hatchery fish spawn with wild fish to varying degrees throughout the Columbia River Basin. In some cases we have virtually no rigorously collected samples to indicate what percentage of the wild spawners are from a hatchery. In virtually all cases, even if we knew what fraction of spawners were hatchery fish, we do not know to what extent those hatchery fish are successful at spawning, or even if they were successful at all. The foundation of the most basic population analysis for any fish stock involves counts of spawners and recruits per spawner. When dealing with wild fish that mix with hatchery fish on the spawning ground, ignorance about the number of hatchery fish and their reproductive success means that all estimates of recruits per spawner are compromised. Without widespread quantitative estimates of hatchery spawning contributions and more selective estimates of relative reproductive fitness of hatchery fish, our analyses (and for that matter anyone's quantitative analyses of salmonid populations) are highly uncertain. For instance, all of the numbers reported thus far assume that there is no reproduction from the hatchery fish. If instead one assumes that a hatchery derived spawner's reproduction is equal to that of wild fish, an extreme assumption for illustrative purposes only, then the average lambdas for the 41 true wild fish from the index stocks where some information is available on the numbers of hatchery fish drops from 0.95 to 0.62 and the magnitude of improvements in lambda needed for recovery skyrockets to over 100% in a large proportion of these cases.

A second sampling uncertainty is the magnitude of sampling error in all fish counts. All scientific measurements include some "observation error." Since many of the run reconstructions for salmonids entail counts of redds (nests) per kilometer of stream that are then converted into estimates of female spawners per kilometer of stream (by some multiplier fraction) and then extrapolated to an entire spawning population, there is clearly great opportunity for an accumulation of observation error. From a scientific point of view, this is acceptable. However, what is *not* acceptable is the fact we have no systematic estimates of how large the observation error in this process is. It was the recognition of this fact that motivated us to design an extinction and population trend analysis that is relatively immune to sampling error. However, as the region moves towards actually attempting management actions and assessing their effectiveness it will be necessary to pay much closer attention to quantifying sampling error for each ESU – which will vary depending on the life history, ESU, lifestage, and watershed. This report produces an analysis of what is called "detectability," the likelihood of detecting an

increase in lambda of some fixed percentage within ten years given the observed variability in the past record of spawner counts. This detectability analysis could be very useful in designing management experiments, but only if it were better informed by estimates of sampling error.

The Need for Management Experiments

A summary of the available data suggests that in the short term, there is little hope of obtaining reliable indicators of the likely efficacy of any of the management options being considered. For example, even for the best studied ESU, the Snake River Spring/Summer Chinook salmon, there is good evidence that dam breaching would increase lambda, but not enough evidence to say by how much. Similarly, there is some evidence that habitat improvements might increase lambda for this ESU, but not enough evidence to calculate a likely percent increase. In order to offer more focused guidance, additional research is needed on quantitative relationships between habitat improvements, dam breaching, and hatchery changes and recruits per spawner.

The data clearly indicate the risks if things stay the same. In addition, we can monitor and see how well lambdas improve following management. However, decision-makers will not be able to turn to existing data to receive a prescription of exactly what actions will reap particular benefits. Instead, the way to interpret the results of this report is to realize that for low lambdas and high extinction risks there is little “wobble room,” and the situation is close to “one must do everything.” If lambda does not reflect such a severe decline and only small improvements are needed, then there may be the potential to choose among different options. Importantly, even the smallest rate of decline means something must be done, and it can be extremely difficult to improve lambda by as little as 1%. The point is that the “worse” (or lower) lambda gets, the less opportunity there is for trying to choose among options and the more discussion should revolve around doing everything. Where exactly the switch between “choices and options” versus “do everything” occurs is a policy decision.

D. SYNTHESIS OF FINDINGS AND RECOMMENDATIONS

Rates of population decline and extinction risks vary widely across the Columbia River Basin, suggesting that management needs vary in accord with these different levels of risk. Most imperiled are Upper Columbia Spring Chinook, Middle Columbia steelhead, Upper Columbia steelhead, and Upper Willamette steelhead ESUs.

The amount of improvement in recruits per spawner that is required to mitigate risks can be modest (less than 1%) or quite large (as high as 65%). When needed improvements are modest there are probably management options, but when needed improvements are large there is little room to be selective about what actions are taken. We must do everything possible to increase recruits per spawners before it is too late. A lambda of 0.9 means that in less than 7 years a population is likely to be reduced to half its current level.

Reductions of harvest represent an easily identified mechanism for improving recruits per spawner in a few ESUs (Lower Columbia Chinook, Upper Willamette Chinook, and Snake River

Fall Chinook). In other ESUs we lack data for making confident quantitative predictions about the likely effects of any particular management action. This is even the case for the much-studied Snake River Spring/Summer Chinook salmon where risks are substantial and the need for action is striking (particularly if one factors in the recent declining trend in recruits per spawner). Although there is some evidence that dam breaching is necessary for mitigating the extinction risk faced by Snake River Spring/Summer Chinook salmon (especially given the lack of evidence that needed improvements can be made by non-breaching management actions), it is highly unlikely that dam breaching alone will recover these populations. Hence, even in this most-studied of all cases, actions will be predicated on uncertainty. But what is not uncertain is the substantial rates of decline for Snake River Spring/Summer Chinook salmon and even worse rates of decline for several other ESUs.

In summary, the scientific uncertainty surrounding the likely outcome of everything but harvest reductions is not an argument for inaction, especially given the high risks faced by several ESUs. Quite the contrary. This level of uncertainty is, however, an observation that the public and policy makers should be aware of. From a scientific viewpoint the ideal action is rapid, targeted management action with effective monitoring programs. Secondly, establishment of quantitative links between management actions and salmon productivity are obviously a priority area for research. The region has suffered from an inattention to standardized reporting of data and analyses at a large scale and as a result currently lacks the scientific information required to make quantitative assessments of management scenarios. It is imperative that this last point be emphasized to the public and policy makers: collectively we have failed to manage Columbia River Basin salmonid populations and are now forced to undertake management actions as experiments, accepting that some will fail, but if they are properly designed, we can learn from our mistakes.

**A STANDARDIZED QUANTITATIVE
ANALYSIS OF RISKS FACED BY
SALMONIDS IN THE COLUMBIA RIVER
BASIN**

This is a preliminary draft made available to groups wanting to preview the material that is basis of the upcoming *NWFSC, American Rivers, Trout Unlimited* jointly sponsored technical workshop 29 March, 2000.

The final DRAFT DOCUMENT will be completed April 5th, and placed on the CRI website for comment. In this first draft document the references have not been completed and some appendices may be missing.

WHAT IS THE PURPOSE AND SCOPE OF THIS DOCUMENT?

The CRI has undertaken a broad analysis of all 12 federally-listed ESUs in the Columbia River Basin. One major purpose of this analysis is to provide: 1) a standardized format for data reporting and analyses, and 2) standardized measures of how well ESUs and populations are doing (with respect to extinction risk and population growth). This is a preliminary analysis that will be revised to make methods and reasoning clearer. This document is intended to provide policy-makers with useful information regarding their options for the entire Columbia River Basin when thinking broadly about needs and options (as opposed to considering one ESU at a time). Key points regarding this document are:

- 1) This is the first time that all Columbia River Basin ESUs or populations have been analyzed in a standardized manner.
- 2) One main purpose of these analyses is to estimate, in relative terms, how much improvement in recruits per spawner will be required to produce acceptable rates of population growth.
- 3) The baseline data being analyzed begin with brood year 1980 and extend through brood year 1994. Because the data extend only until brood year 1994, some of the most recent management actions (such as recent harvest reductions) have not had time to be expressed in the data.
- 4) In addition to the standardized analyses of all 12 ESUs, analyses for the Snake River ESUs (notably Fall Chinook salmon and Spring/Summer Chinook salmon) have been updated.
- 5) Among the changes from previous CRI documents are: new data added, different extinction thresholds used, an improved extinction analytical approach adopted, and matrices estimated in multiple ways.

Please e-mail comments to Peter.Kareiva@noaa.gov – with a subject line of: ESU12 COMMENTS. The document will be revised April 5th; hence comments must be received by April 2nd, in order for them to be included in the final DRAFT DOCUMENT.

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I. Introduction to Problems Faced by Columbia River Basin Salmon

Since 1991 twelve salmonid Evolutionarily Significant Units (ESUs) in the Columbia River Basin have been listed as threatened or endangered under the Endangered Species Act. Human impacts on the river basin and the species inhabiting it have led to dramatic declines in numbers of anadromous fishes and in the quality and quantity of habitat available to them. It is widely recognized that large sacrifices need to be made in many areas if these salmon are to be recovered (NRC 1996), but there is a lack of quantitative analysis for guiding what sacrifices might be most effective, or which ESUs and populations are at the greatest risk. This document presents the sorts of analyses needed to conduct basin-wide planning, but in no way is this document the final word; indeed some of the key findings pertain to subsequent analyses and data syntheses that need to be initiated as soon as possible.

I.A. Physical Setting and Historical Impacts

The Columbia River Basin covers about 250,000 square miles (an area nearly the size of Texas), in seven U.S. states and British Columbia, stretching from the Pacific Ocean to the Continental Divide. Within its boundaries are found wet and dry temperate coniferous forests, arid shrubland and grasslands, true desert, and alpine areas.

Returns of salmon and steelhead to the Columbia River basin before Euro-American influence have been estimated to range from 7-8 million (Chapman 1986) to 10-16 million (NPPC 1986) per year. Population declines appear to have begun in the late 1800s due to harvest pressures. Habitat degradation and loss, as well as blockage or impediments to fish passage from the construction of dams throughout the basin exacerbated these declines. Current returns of adult salmonids to the entire basin are about 1 million fish per year; approximately 80 percent of these fish are of hatchery origin.

Significant non-tribal harvest pressures began with the establishment, in 1866, of the first of forty commercial canneries on the Columbia. Peak in-river harvest may have been as high as 3-4 million fish (Chapman 1986), and the catch of spring chinook salmon began a steady decline after the 1870s, triggering switches in harvest to other species and runs. By the 1940s, the total catch of salmonids from the Columbia River was about half of that in the late 1800s (WDFW, ODFW 1994, cited in NRC 1996). Both harvest methods and rates have been regulated in this century, and several ESUs are currently subjected to only ceremonial and subsistence treaty harvest. Other stocks are more heavily impacted by both ocean and in-river fisheries.

The first salmonid artificial propagation program in the region was established in 1877 on the Clackamas River. Today, there are more than 80 hatcheries in the Columbia River Basin, releasing over 100 million juveniles annually (CBFWA 1990, NRC 1996).

Although hatchery fish have played a role in sustaining run sizes, artificial propagation programs have a variety of potentially detrimental impacts on natural populations. Most directly, the presence of hatchery fish in mixed fisheries has led to harvest rates that cause overfishing of wild populations. In addition, the formerly widespread practice of using non-local stocks has contributed to a loss of natural patterns of genetic variation both within and between populations (NRC 1996). Other potential effects of hatchery releases on wild populations include a reduction in the average fitness of wild populations due to interbreeding between wild and hatchery fish, increased competition due to the increased numbers of fish in the system, and predation on juvenile wild fish by the hatchery released fish.

Habitat degradation has been a long-term problem across the Columbia River Basin, and stems from many human activities. Grazing, logging, agriculture, mining and urbanization have had a wide variety of negative effects on riparian and riverine systems including increased runoff and erosion, higher sedimentation, increased channelization of rivers and streams, higher water temperatures, fewer pools and less cover in stream systems, and extensive alteration of native plant communities. These habitat impacts began early in the history of Euro-American settlement of the region, with livestock in the basin reaching their highest numbers before 1900, and the first sawmill constructed in 1827. While grazing has been reduced on public lands in recent years, riparian habitat conditions on public rangeland remain generally poor (GAO 1988). Logging continues to be a leading industry throughout the Pacific Northwest, with over 10 billion board feet harvested annually in Oregon and Washington through 1990. (Timber harvests on public lands have decreased in recent years). Habitat degradation and alteration does not stop at the river mouth, either. An estimated 65% of tidal swamps and marshes in the Columbia estuary have been lost due to diking and filling (Thomas 1983).

Construction of dams was the final insult to a system already taxed by human alteration. A total of 27 large dams were built between 1930 and 1975 on the Columbia and Snake Rivers within the historic range of anadromous salmonids. In addition, over one thousand smaller dams, some without fish passage facilities, are currently maintained on tributaries to the Columbia and Snake Rivers. In total, about 55% of the area and 31% of the stream miles formerly available to anadromous fishes have been blocked by dam construction (NRC 1996). Dams have also inundated important mainstem spawning habitat. Currently the healthiest (fall) chinook salmon population in the basin spawns in the Hanford Reach, the only accessible unimpounded stretch left on the mainstem of the Columbia River. Out-migrating juvenile fish must negotiate the reservoirs (and the non-native predators found there) and the dams on their way to the ocean. Juvenile fish pass the Snake and Columbia River dams through bypass systems, over the spillways and through turbines. Most effort in the past decades has focused on increasing juvenile survival through the hydropower corridor with improved collectors, bypass systems and transportation.

Finally, salmon populations also appear to be affected by variation in ocean conditions associated with short and long-term climatic fluctuations. In particular, salmon

production appears linked to decadal-scale shifts in average sea surface temperatures (the Pacific Decadal Oscillation; Francis and Hare 1994, Hare et al. 1999). These temperature changes affect phytoplankton production, which in turn affects zooplankton abundance (Broder and Ware 1994, Sugimoto and Tadokoro 1997). Phytoplankton production appears to be very important for juvenile salmonid survival in the ocean. The variation in survival linked to these climatic conditions will contribute significantly to recruitment variation, although the exact effect of marine mortality varies among salmonid species (Bradford 1997).

I.B. Evolutionary Significant Units

Nearly as complex as the myriad factors affecting salmonid population trends is determining the appropriate unit for conservation. Amendments to the Endangered Species Act in 1978 defined a species as "any subspecies of fish or wildlife or plants, and *any distinct population segment of any species* of vertebrate fish or wildlife which interbreeds when mature" (emphasis added).

NMFS considers a population or group of populations "distinct" and hence a "species" for purposes of the ESA if it represents an evolutionarily significant unit (ESU) of the biological species. A population must satisfy two criteria to be considered an ESU: first, it must be reproductively isolated from other conspecific population units, and second, it must represent an important component in the evolutionary legacy of the species. In defining ESU boundaries, genetic, demographic, life history, morphological and geographic information was considered (e.g., NMFS 1991 a, b, c, d, 1996, 1998). The twelve ESUs in the Columbia Basin listed under the Endangered Species Act that we consider here are listed in Table I-1.

Management of these listed ESUs is made particularly challenging not only by the wide array of habitats that they use, but also by the extreme variation in life history characteristics shown. Salmonids in the Columbia River basin spawn in the main river channel of the Columbia and Snake Rivers, in small streams and tributaries and in lakeshore gravels. Not only are both ocean and stream-type fish represented, but at least two ESUs include individuals that are not anadromous at all. As a result, juveniles or adults are migrating to or from some part of the basin in all months of the year. Therefore, management actions aimed at improving the situation for one ESU must be evaluated for their effects on other ESUs as well.

Table I-1. Salmonid ESUs listed under the Endangered Species Act in the Columbia Basin.

Species	ESU	Status
Chinook	Lower Columbia River	Threatened
	Upper Willamette River	Threatened
	Upper Columbia Spring-run	Endangered
	Snake River Spring/Summer-run	Threatened
	Snake River Fall-run	Threatened
Steelhead	Lower Columbia River	Threatened
	Upper Willamette River	Threatened
	Middle Columbia River	Threatened
	Upper Columbia River	Endangered
	Snake River	Threatened
Sockeye *	Snake River Basin	Endangered
Chum	Columbia River	Threatened

* Because Snake River Sockeye are near-zero in abundance, the risk of extinction is clear. As a result, we did not include this ESU in any of the quantitative analyses presented in this document.

II. Goals and General Considerations

II.A. Goals

This paper has three important goals:

- **Address technical aspects of the Dennis extinction risk analysis with respect to available salmon data.** In Section III we present tests of the assumptions of the Dennis model (Dennis et al. 1991) for each of the ESUs investigated. In addition, we provide a refinement of the model that is robust to sampling error. Finally, we outline an approach to address the issue of non-linear trends that appear to be present in some stocks.
- **Provide a standard assessment of extinction risk for all listed ESUs in the Columbia River Basin.** In section IV we present results from our extinction risk analysis of eleven ESUs to allow comparison of extinction risk faced by these groups. We examine both the likelihood of stocks within each ESU reaching absolute extinction, and the probability of the ESU as a whole, or stocks within an ESU, experiencing significant declines. We also present the necessary increase in population growth rate (or recruits-per-spawner) to reduce this risk to 5% in 100 years.
- **Provide more detailed analyses for those ESUs with sufficient data to support such an analysis.** Section V includes revised analyses for the Snake River spring/summer chinook. These analyses are based on the most recent spawner data, provided to us within the last two months. We also provide analyses for Snake River Fall Chinook in this section. An update of this document will include analyses of Upper Columbia steelhead and spring chinook, developed during the QAR process.

In addition, in Section VI, we present a stochastic simulation model for extinction risk that is in the process of being developed.

II.B. General Considerations

We have applied a standardized analysis to the ESUs in the Columbia River Basin. Each ESU obviously has its own suite of characteristics and considerations. However, there are several general factors that should be considered while interpreting or applying these analyses.

II B.1. Simple analyses

This paper presents a chain of simple analyses, addressing questions pertaining to salmon risk and risk management. The analyses we conduct are supported by the available data, which in many, if not most, cases is restricted to counts of spawners or

redds. Overparameterizing models or analyses, by including factors about which there is little information, can degrade the performance of more complicated analyses (Burnham and Anderson, 1998). In fact, due to the difficulties associated with estimating parameters, simple models can perform as well as more elaborate or detailed analyses (e.g. Ludwig and Walters, 1985). For these reasons we use available data in a sequence of simple analyses as a chain of arguments in order to simplify the complexity of nature (Caswell, 2000). As technical teams develop recovery goals and apply VSP standards to particular ESUs, the CRI will tailor its analyses to these more specific venues. For instance, future analyses will assess carrying capacity and long-run population levels as well as population growth rates. These estimates will necessarily involve additional kinds of data and experiments; reliably estimating carrying capacity requires map-based assessments of habitat amount and quality, as well as quantitative links between habitat attributes and some measure of salmon population productivity.

II.B.2. Stocks and Populations

Populations within ESUs in the Columbia River will eventually be defined using a combination of genetic, life history, demographic and other characteristics (VSP paper, McElhany et al. 1999). This process is complete for Upper Columbia River Spring Chinook and Steelhead, but has not begun for any other Columbia River ESUs. Since populations have not been delineated using biological criteria, we have used data for geographically defined stocks. In some cases, these stocks will correspond to biological populations. However, in others, a single stock may consist of more than one population, or several stocks may all belong to a single population. In these cases the risk metrics we provide may be inaccurate. For instance, analyzing a small stream as an independent population when it is actually part of a much larger population, receiving migrants from other streams or rivers in the population, may overestimate extinction risk. Similarly, lumping several streams together as a population, when they are in fact independent, may cause the extinction risk for some of the smaller independent units to be underestimated. Therefore, when populations are formally delineated estimates of extinction risk may change.

II.B.3. Time period analyzed

We used data from brood year 1980 to the present for both extinction analyses and Leslie matrices. Although in some cases data are available before 1980, we selected a 1980 starting date several reasons. First, the four lower Snake River dams were completed in 1975, and the full complement of turbines installed by 1979. Additional engineering changes to these and other mainstem Columbia River dams were completed by the early 1980s. Thus, prior to 1980 the hydropower system was in a state of flux, while current operations have more or less been in place since 1980. In addition, a major shift in oceanographic conditions, accompanied by substantial decreases in marine primary production (Mantua et al. 1997), coincided with the completion of the Federal Columbia River Power System in 1975 (corresponding to outmigration year 1977). Also in 1975, completion of the Libby and Mica Dams in the

upper reaches of the Columbia basin nearly doubled storage capacity in the river at the same time that climate shifts decreased annual average run-off. Finally, in some cases the quality of early data is not uniform across ESUs (Zabel and Williams in press). Thus, by using more recent data we eliminate some, though certainly not all, problems with differences in data quality among ESUs. By restricting our analysis to the time period after 1980, we are, in effect, asking the question: **if trends continue as they have for the last twenty years, what is the likely status of the ESU (or stock) at some future date, relative to other ESUs (or stocks)?**

II.B.4. Risk metrics

We consider three “extinction” metrics: probability of absolute extinction, probability of a 90% decline in abundance and probability of a 50% decline in abundance. Measures of decline in abundance are critical for two reasons. First, it is often difficult to obtain an estimate of the number of fish in the entire population. In these cases, an accurate estimate of the probability of reaching an abundance threshold is impossible to achieve. (This is particularly true for ESU-level data, since often only sub-samples are available. We did not consider any abundance threshold metrics at the ESU-level for this reason). Second, in some cases, population numbers are currently relatively high, but the overall trend is declining. Probability of reaching a threshold in these cases is relatively low, even though the population is clearly at risk. A measure of decline, therefore, offers a measure of risk that is independent of absolute abundance.

We have not considered the probability of reaching an abundance threshold greater than absolute extinction in this document for two reasons. First, as discussed above, data are often not available at the ESU level to determine the risk of reaching a particular population size. Put another way, an extinction threshold of 1-fish is the only extinction threshold that has the same biological meaning regardless of which index stock or population is being examined. Second, a primary goal of this work was to provide a standardized and comparable assessment of risk across the stocks in the Columbia River Basin. At the stock level, the geographic or stock units for which we had data were often not comparable. For instance, comparing the probability of chinook populations of small streams in the lower Columbia River reaching 250 individuals with populations in the Yakima River compares two systems of very different potential capacities. This presentation of the risk of absolute extinction alone does not reflect a policy decision about acceptable population levels, nor does it indicate that CRI scientists are ignoring the possibility that depensation occurs in these populations. Instead, it is the only level that we can be certain is biologically meaningful across all systems. As populations are delineated according to biological criteria, and depensation thresholds are better defined, we will be in a better position to apply other abundance thresholds. Obviously, the extinction threshold we have used – absolute extinction – is dire, and estimates of risk must be considered in this light.

A final critical metric is “lambda”, or the annual rate of population change. Lambda, more than any other parameter, summarizes the likely fate of a population. In general, “managing for lambda” is a reliable way of managing for a species viability and

productivity (Caswell 2000). Later in this document we calculate the lambda required to reduce certain risks of extinction to less than 5% over a 100 year time period. The lambda's that satisfy this requirement are quite large, and would represent thriving salmonid populations. A second value of lambda as a metric is that one can estimate 95% confidence intervals of lambda from previous data, and hence ask whether future population trends indicate significant improvements or deteriorations in the situation.

II.B.5. "Acceptable" risk

We have calculated the change in population growth rate necessary to reduce the risk of absolute extinction for individual stocks to less than 5% in 100 years. This value was chosen based on IUCN standards (1994) for inclusion of species in its Red Lists. (A species with a ten percent or greater risk of absolute extinction in 100 years is considered to be *vulnerable*, the lowest category of concern). Again, this level is not a policy statement about jeopardy or recovery standards. Rather, it is a convenient and internationally accepted standard against which to gauge necessary improvements.

II.B.6. Change in annual population growth rate vs. change in recruits-per-spawner ratios.

In general, we have presented results in terms of annual population growth rates (lambda, *I*). This quantity (the average rate of change in the population size per year) is not equivalent to the commonly used recruits-per-spawner ratio (rate of change in the population size per generation). However, lambda and recruits-per-spawner have the following relationship:

$$I = (\text{average } R/S) * (\text{mean return time})^{-1} * e^{\frac{1}{2}s^2}$$

or

$$I = (\text{average } R/S) * (\text{constant})$$

$$\text{where } s^2 = \text{var}((R/S)_t)$$

Therefore, the percent change in lambda is the same percentage change required in recruits-per-spawner to reduce extinction risk to a specified level.

III. Dennis Extinction Risk Analysis – Tests and Refinements

III.A. Are extinction analyses useful?

Estimates of extinction risk are standard “tools of the trade” in conservation biology. But these very same estimates have come under criticism for a wide variety of reasons. Recently, several mathematical biologists have cautioned that point estimates of extinction risk typically have such huge confidence intervals (e.g., Ludwig 1999; Frieberg and Ellner 2000) that the estimates become “meaningless”. In some cases these reports of huge confidence intervals are unduly pessimistic because they apply only to large time horizons, to very sparse data sets, and to less-than-the-best analytical methods. But, in general, large confidence intervals are common. But as Caswell points out, to say that uncertainty makes the results “meaningless” creates an erroneous distinction between results and uncertainty, “*uncertainty does not make the results meaningless, the uncertainty is part of the results*” (Caswell 2000). In addition, we need to stop and ask how one uses estimates of extinction risk. The simplest use is to compare stocks, to compare ESUs, to compare management options, and evaluate relative risks. For these uses the confidence interval is less of a problem, because one is assessing relative risk and not absolute risk. But secondly, one needs to consider the alternative to calculating an extinction risk metric. The alternative is either randomly ranking populations with respect to risk, or to use “expert guesses” (which are very hard to standardize). The value of standardization is so substantial that we have adopted the modified Dennis approach as the minimum standard for analyses across all ESUs, well aware that there is uncertainty surrounding each estimate. At least the assessment of risk is explicit, repeatable, and responsive to the addition of new data. Simultaneously, we have undertaken several theoretical studies to understand in what direction its biases might be, and under what circumstances those biases could be large. Others are pursuing similar analyses, and results thus far suggest that many simple viability models, which are clearly over-simplifications, still perform surprisingly well (Meir and Fagan 1999, Fagan et al. 1999, Caswell 2000). Of course, to properly apply these methods the key assumptions underlying them should be tested with the data being analyzed so that sources of error are clearly exposed.

III.B. A simple extinction model with minimal data requirements – the Dennis approach

In 1991, Dennis et al. published an extremely important and influential paper that outlined an approach to quantifying extinction risk with census data as opposed to detailed demographic studies. Dennis et al. has found wide application, and has been extended and compared to more complex models in several different research publications (Meir and Fagan 1999, Morris et al. 1999). The Dennis model approach reflects the strategy of estimating statistical properties from a time series and then projecting the population forward using those statistical properties. Suppose that one

had population counts for a particular species (see Figure III-1). One could then analyze the distribution of N_{t+1}/N_t and project the population into the future by a stochastic simulation such as:

$$N_{t+1} = N_t * \mathbf{q}(t, N) \quad [3.1]$$

where $\mathbf{q}(t, N)$ is the distribution of N_{t+1}/N_t and might be a function of time and population size. The projected population trajectory would be one of many possible trajectories since this is a stochastic simulation. By running the simulation a large number of times, one could determine probabilities of the population going extinct in certain timeframes. Notice that with this particular method for a viability analysis, no specific population dynamics model is fit to the data, but rather the statistical properties of N_{t+1}/N_t are determined and these are used to parameterize a stochastic simulation [3.1] which is then used to explore the distribution of times to extinction.

For a wide variety of species, including species with strong age-structuring such as salmon, it has been observed that the trajectories of the total population size often have particular statistical properties (Dennis et al. 1991). Namely, the ratio of N_{t+t}/N_t is lognormally distributed with variance $\mathbf{s}^2 t$ and mean $\mathbf{m}t$, where N_t is the total population size at time t and \mathbf{m} and \mathbf{s}^2 are parameters that characterize the rate of population decline and its variability. These statistical properties of N_{t+t}/N_t have a strong theoretical foundation. Studies on stochastic matrix models have shown that the total population size of age-structured populations with year-to-year variance in survivorships and fecundities and no-density dependence should behave as a stochastic markov process of exponential increase or decrease with process error:

$$N_{t+t} = N_t \exp(\mathbf{m}t + \frac{\mathbf{e}}{2}t) \quad \text{where} \quad \mathbf{e} \sim N(0, \mathbf{s}^2) \quad [3.2]$$

This implies that $\frac{N_{t+t}}{N_t} \sim \Lambda(\mathbf{m}t, \mathbf{s}^2 t)$, i.e., is lognormally distributed (Dennis et al. 1991).

Note that the stochastic model (equation 2) is based on the assumption that $\mathbf{q}(t, N) = \mathbf{q}$, that is the distribution of N_{t+t}/N_t is neither a function of time or population size. In salmon terms, this is equivalent to assuming that the R/S ratios are not changing in time and are not density-dependent.

The parameter, \mathbf{m} , is the estimated instantaneous rate of decrease (or increase) for the underlying stochastic process that fits the observed time series data. The predicted \hat{I} of the data (the mean rate of decline) is:

$$\hat{I} = \exp(\hat{\mathbf{m}} + \hat{\mathbf{s}}^2 / 2) \quad [3.3]$$

Why is \hat{I} greater when \hat{s}^2 is greater? The underlying process is assumed to be $N_{t+\tau} = N_t \exp((\mathbf{m} + \mathbf{e})\mathbf{t})$. This is a non-linear function and when there are "good" years ($\mathbf{e} > 0$) this creates a bigger relative increase than an equal in magnitude "bad" year ($\mathbf{e} < 0$). The overall result is that good years are more common when \mathbf{s} is bigger and consequently \mathbf{I} is larger. Note, however, a bigger \mathbf{s} will also increase the likelihood of hitting 0 by chance. If a time series satisfies the key assumptions underlying our modified Dennis approach, then \mathbf{I} and \mathbf{s}^2 correctly characterize the population behavior being exhibited by the data.

III.C. Are the assumptions of the Dennis model met?

The Dennis approach to estimating extinction risk entails several critical assumptions and restrictions:

- I. Population counts must be an exhaustive survey of the population or a fraction thereof so that the timeseries (N_t N_{t+1} N_{t+2} ...) is indeed a Markov process where N_{t+1} is directly related to N_t .
- II. The variability estimated by the modified Dennis approach is a measure of environmental variability and not sampling error.
- III. The variance in $N_{t+\tau}/N_t$ increases with tau (τ) the time increment over which the change is calculated.
- IV. The yearly rates of population growth (i.e., N_{t+1}/N_t) are lognormally distributed.
- V. Although the populations themselves may be increasing or decreasing (i.e., show a trend), there should be no trend in the rates of decline or increase (such that the rate of decline is getting progressively worse or better).
- VI. Over the range of population sizes examined, the rates of population change are assumed to be independent of the density of fish.

Assumption 1: Are population counts an exhaustive survey of the population, such that the time series is a Markov process?

Spawner counts are not an exhaustive survey of the population, or a fraction thereof. In addition, spawner counts at time $t+1$ do not bear a direct relation to spawner counts at time t . Therefore, applying the Dennis model in the standard fashion to spawner counts will grossly overestimate the variance, and therefore overestimate extinction risk. We present a refinement of the Dennis model in Section III.D. to deal with this violation.

Assumption 2: The variability estimated by the modified Dennis approach is a measure of environmental variability and not sampling error

Although recent numerical work has indicated that extinction risks estimated by the Dennis et al approach are robust to modest amounts of observation error (on the order of 25%, see Meir and Fagan 1999), the observation error in run-reconstruction data is likely to be extremely large, probably larger than 25%. This is especially evident when one realizes that often there are less than ten spawners (maybe even as few as one or two spawners) as the denominator of recruit per spawner ratios; a miscount of only one or two fish at such low population sizes can easily yield errors of 50 to 100%. Fortunately, however, s^2 can still be estimated from stage-specific data (such as spawner counts) by using a modified estimation procedure (Holmes 2000) that reduces the problem of inflated estimates of environmental variation due to sampling error. We present this refinement in Section III.D.

Assumption 3: Linear increase through time in variance of $N_{t+\delta}/N_t$

The assumption upon which our estimate of s is based, is that the variance in $\log(N_{t+\delta}/N_t)$ increases linearly with t . For each ESU and population analyzed, we plot the change in variance for $N_{t+\delta}/N_t$ versus time. Figure III-2 shows the variance of $\log(N_{t+\delta}/N_t)$ versus t for our lumped ESU population data. The variance is markedly linear for 9 of the 12 ESUs; the variance for the remaining three is near zero. These latter three situations correspond to cases where there is virtually no variance in the rate of population change (e.g., straight declines at a constant rate). Only the Columbia River Chum data set strongly violates the non-linearity assumption. The reason the Columbia River chum violates the linearity assumption is evident from inspecting its population behavior (see Figure III-3) – its population shows striking periodic behavior during the interval sampled.

In general, the linear increase in variance indicates that the estimation procedure and diffusion approximation is likely to succeed. For those situations that markedly violate this linear assumption, there is typically evidence of a cycle or periodic population behavior, such that instead of steadily increasing through time, the variance levels off or drops due to a return to previous conditions.

Assumption 4: Lognormal rates of yearly population growth N_{t+1}/N_t

Frequency distributions of N_{t+1}/N_t generally satisfied the assumption of a lognormal distribution. Plots of these distributions for several ESUs and individual stocks are illustrated in Figure III-4. This assumption is likely to be satisfied for most time series of population growth rates because these growth increments will generally be the product of many independent factors operating on reproduction and survival.

Assumption 5: A population's rate of change may fluctuate, but it is not getting progressively better or worse in a predictable fashion.

A key technical challenge is appropriately addressing the possibility that trends in environmental degradation are linked to increasing declines in annual rates of

population growth through time. Such declines would be far more serious than simply declining populations, because they would imply that not only are populations declining, but also that the rate of decline is accelerating. We are addressing this issue in two ways. First, we graphically display the time series used in our analyses so that any bias inherent in the extinction analyses resulting from a trend is apparent. Second, we are developing new analytical tools that can formally incorporate trends using formal statistical models (Burnham and Anderson 1998).

Simple plots of spawner population abundance (Figure III-5) and annual growth rates (Figure III-6) suggest that while there are trends for some of the ESUs or stocks examined, trends are not apparent in the majority of cases. In particular, plots of the temporal pattern in $\log(N_{t+t}/N_t)$ indicate no consistent upward or downward trends from 1980 onward for all but Upper Columbia Spring Chinook salmon and Snake River Spring/Summer Chinook salmon (Figure III-6). However, time scale can affect the appearance of a trend. If one includes data prior to 1980 (Figure III-7), the Snake River chinook salmon does not show as unambiguous a trend as the post-1980 data might indicate. To be conservative however, it is best to conclude that the extinction risks estimated by the modified Dennis approach are underestimates for Upper Columbia Spring chinook salmon and Snake River spring/summer chinook salmon. This underestimation arises because the data suggests that the decline is becoming progressively steeper with time.

Since trends are not evident across all ESUs, and the impression of a trend varies with the time scale examined, a formal means of identifying trends in recruits per spawner is needed. One approach is the Dennis and Taper (1994) likelihood ratio approach in conjunction with bootstrap simulations we use to test for density-dependence (see Assumption 6 below). Another approach might be to formally propose a new diffusion approximation that includes a trend in the drift parameter. We are currently working on such an approach; unfortunately this is not a straightforward problem. One major problem is that density-dependence and a declining trend in recruits per spawner based only on yearly counts (not density) can produce nearly identical and statistically indistinguishable patterns of population decline. Moreover, moving toward this complicated analysis means that some of the transparency of our simple analytical process is lost.

Until we or other scientists in the region perfect an approach to trend modeling that is clear and unambiguous, we advocate inspecting plots such as those shown in Figures 6 and 7, and using that examination to color the interpretation of the simple extinction results. To repeat the example from above, there may indeed be a declining trend in recruits per spawner for spring/summer chinook salmon in the Snake River, and that hence the results of those extinction analyses may be underestimates and hence too optimistic.

Assumption 6: Density-independence

The Ricker function and its many modifications have enjoyed a long history as the premier population growth models employed in fisheries biology. The Ricker model

assumes that the log of the rate of recruitment per spawner decreases linearly as spawner density increases. A critically important parameter for assessing extinction risk is the per capita production of recruits when populations are low (near extinction), which can be estimated from a Ricker model as the intercept of the linear regression relating natural log of “recruits per spawner” to the number of spawners. In practice, estimates of this parameter based on a Ricker function are biased toward producing unduly optimistic portraits of the future for populations (Ginzburg et al. 1990), because they assume that there will be greater recruitment as the number of spawners decreases. It is worth noting that to date, most extinction risk analyses applied to salmonid populations have relied upon density-dependent models (e.g., Emlen 1995; Ratner et al. 1997). It is not clear how much the assumption of density-dependence may have positively biased the probability of persistence estimated in these particular simulation efforts.

To assess assumptions regarding density-dependence, the time series of spawner and recruit counts were subjected to a likelihood ratio test for density dependence (Dennis and Taper 1994). This test, which is far more robust than alternative methods, compares the likelihood of the data if the underlying model is a stochastic logistic (density-dependent) model (allowing for either compensation or depensation) to the likelihood of the data if the underlying process is a stochastic exponential (density-independent) model. The distribution of the test statistic is generated by parametric bootstrapping. If the calculated test statistic is greater than 95% of the randomly generated values, the null hypothesis of density-independent growth is rejected. This test for density-dependence is provided for each ESU, and stock within an ESU for which long time series were available.

Of the 11 stocks tested only one, Upper Columbia River Spring Chinook, showed any type of density dependent population regulation (Table III-1). All other stocks, declining and increasing, large and small population size, showed no relationship between numbers of spawners and the population growth rate, I . The test statistic (T_{12}) for the Upper Columbia Spring Chinook fell well above the 95%tile of simulated T-statistics, indicating that the density dependent model was a better fit to the population time series. In this case, the population was declining ($a_2 < 0$), but the density dependent coefficient was positive ($b_2 > 0$), implying that for small population sizes the population growth rate is less than at larger population sizes, i.e., depensation. Depensation is thought to occur in populations with small numbers of reproducing individuals for a variety of reasons, primarily reduced mating probability due to spatial and temporal segregation of individuals on the breeding grounds. Depensation is one of several demographic hazards of small population sizes and is of interest in conservation management of species at risk. Depensation-like density dependence has been hypothesized for salmon, however no good data, or predicted thresholds for when it may become an issue, exist. Unfortunately, for declining stocks, depensation-like effects can also be attributed to a downward trend in I . For example, if the environmental conditions are deteriorating steadily such that population growth rates are declining, then decreasing population sizes will be observed concomitant with decreased population growth rates. The problem then becomes one of distinguishing

between a trend in I with time or with population size (N). Figure III-8 shows regression models for I against run year and population size. Both models are significant ($p < 0.05$), but the dependence of I on run year explains more of the variance in the data than does its dependence on N ($r^2 = 0.77$ vs $r^2 = 0.56$). Thus, it is certain that the Upper Columbia River Spring Chinook show a decreasing population growth rate, but it is not possible to attribute this trend entirely to environmental degradation or compensatory demographic processes, nor potential interactions between both mechanisms.

Table III-1. Test for density dependence in Columbia River and Washington Coast stocks. The parameters $a1$, and $a2$ and $b2$ are fits of the data to exponential and logistic population growth models and the variance around the fits, $var1$ and $var2$, respectively. The test for density dependence compares the T-statistic to the 95th %tile ranking of 2000 simulated population trajectories (the distribution of T_{12} is not known). In only one stock, Upper Columbia Spring Chinook, was the observed statistic greater than 95% of the simulated values.

Stock	$a1$	$var1$	$a2$	$b2$	$var2$	T_{12sqr}	$test95$
LC Ck	0.030	0.046	1.026	-7.00E-06	0.017	3.402	14.571
UC Sp Ck	-0.163	0.050	-0.526	3.20E-05	0.022	14.810	4.437
Sn S/S Ck	-0.034	0.022	-0.080	5.00E-06	0.022	0.055	5.001
Sn F Ck	-0.056	0.016	0.090	-5.10E-05	0.014	0.768	4.517
UW Ck	0.009	0.056	0.155	-1.10E-05	0.051	0.938	8.663
CR Ch	0.054	0.106	0.300	-1.68E-04	0.092	1.488	7.654
LC W Sh	-0.063	0.003	-0.097	2.00E-06	0.003	0.011	6.956
LC S Sh	-0.050	0.008	-0.022	-1.00E-06	0.008	0.002	7.168
MC Sh	-0.120	0.003	-0.228	3.00E-06	0.002	0.911	7.042
UC Sh	-0.061	0.023	-0.111	1.20E-05	0.023	0.117	4.645
Sn Sh	-0.024	0.003	0.004	0.00E+00	0.003	0.030	4.889
UW Sh	-0.074	0.029	-0.120	2.00E-06	0.029	0.165	4.675

III.D. Refinements of the Dennis model

We have made some refinements to the Dennis model in order to deal with violations of two of the criteria or assumptions presented above: population counts do not represent a Markov process, and sampling error is likely to be significant. As mentioned above, we are exploring some formal methods to assess and incorporate trends into the risk assessment.

III.D.1. Estimating total spawner population size

In order to ensure that the time series of population counts used in our analyses did indeed represent a Markov process, we estimated the total population of spawners and potential spawners using spawner counts and age structure data. For individual stocks, we have an estimate of total spawner population size:

$$N_t = \sum_j^{\max \text{ age}} w_j S_{t-j} \quad [3.4]$$

where w_j is the mean number of spawners from spawners at time $t-j$ that are returning in year t or will return (but have not yet). The spawner population includes those that are returning and those that are in the ocean but will return. The weighting function is,

$$w_j = 1 - (\text{frac of a cohort of spawners that have already returned before year } t-j).$$

III.D.2. Estimating variance accurately

Sampling error is likely to be large in spawner data, resulting in an overestimation of variance and therefore, of extinction risk. Fortunately, the variance (s^2) can still be estimated from stage-specific data (such as spawner counts) by using a modified estimation procedure (Holmes, 2000). Briefly, s^2 can be estimated from the slope of y versus t in:

$$\text{var} \left(\frac{\sum_j w_j S_{t+t-j}}{\sum_j w_j S_{t-j}} \right) \text{ versus } t \quad [3.5]$$

where S_t is the spawner count at time t . This is possible because the variance of the weighted sums for stage or age counts is approximately $(s^2 + C_1)t + C_2$ where C_1 and C_2 are unknown constants and C_1 is small if the weighting function, w_j , is correct and/or does not overlap greatly (Holmes 2000). One of the advantages of estimating variance from the slope of this relationship is that the slope is NOT altered by observation error. Similarly, to minimize the influence of observation error on the

estimation of mean rate of population growth, we use the slope of N_{t+t}/N_t versus t . Simulations indicate that this gives estimates of mean rate of population growth that better mesh with observed patterns of population decline than does the standard simpler estimate based on simply the mean of N_{t+1}/N_t .

Because of the modifications we have adopted for estimating instantaneous rates of change and environmental variance, our modified Dennis approach is not likely to founder on the assumption of no sampling error. However, almost all other extinction metrics will be markedly altered if sampling error is large and not properly accounted for in the analyses.

III.E. Extinction Risk Metrics and Data

III.E.I. Extinction Risk Metrics

If we know the rate of population change and the magnitude of environmental variation in that rate of population change, it is straightforward to estimate extinction risks by running a stochastic simulation that has been appropriately parameterized. The observed probability of extinction over a large number of simulations gives an extinction risk metric (also called the probability of extinction). However, if N_{t+t}/N_t is lognormally distributed one can determine the probability of the stochastic simulation going extinct without actually having to run simulations. This is because the stochastic process with lognormal N_{t+t}/N_t can be approximated as a diffusion process:

$$\ln(N_{t+t}) = \ln(N_t) + \mathbf{m}t + \mathbf{e}t. \quad [3.6]$$

The behavior of such diffusion processes is well established and in particular the probabilities and times to reaching thresholds, such as extinction, can be easily calculated. In particular, the probability of reaching a particular threshold N_e from the current population size N_0 within time t_e is

$$G * \mathbf{p}' = \mathbf{p}' * \Phi \left(\frac{-\ln(N_0/N_e) + |\mathbf{m}| t_e}{\mathbf{s}t_e} \right) + \exp(2\ln(N_0/N_e)|\mathbf{m}|/\mathbf{s}^2) \Phi \left(\frac{-\ln(N_0/N_e) - |\mathbf{m}| t_e}{\mathbf{s}t_e} \right) \quad t_e > 0 \quad [3.7]$$

$$\text{where } \mathbf{p}' = \begin{cases} 1, \mathbf{m} \leq 0 \\ \exp(-2\mathbf{m}\ln(N_0/N_e)/\mathbf{s}^2), \mathbf{m} > 0 \end{cases}$$

Another quantity that we use is the probability that the population is 90% lower than its current population size at time t_e . This is (Dennis et al. 1991):

$$\Pr\left(\frac{N_{t+t}}{N_t} < \frac{x}{1}\right) = \Phi\left(\frac{-\ln(x/1) + \left|\sum_i m\right| t_e}{\sum_i s t_e}\right) \quad [3.8]$$

where $x=10$ for a 90% decline. We use the probability of a 90% decline to measure risks in those cases where we may not know what the total population is, but still want to quantify the risks of perilous declines. For example, at the level of an entire ESU, it usually is impossible to know what is the total population for the ESU (since only subsets of it are sampled), but the sampled portions of the ESU could nonetheless give an apt portrait of the risks of severe declines. In some cases we examine probability of a 50% decline (in which case $x=2$ in equation 13). Clearly many healthy populations regularly suffer 50% declines; we use the “50% decline metric” simply to compare the behavior of populations as indicated by their recent history of population changes (not as a measure of extinction likelihood).

III.E.1.a. Extinction Risk Metrics for the ESU level

All ESU-level analyses assess risks in terms of the probability of a 90% decline. However, at the ESU-level, we must deal with the amount of environmental and dispersal-mediated correlation between populations within the same ESU. Populations that are composed of completely independent stocks have lower probabilities of decline than populations that are correlated either via dispersal or environment. We are currently working on methods for teasing apart environmental versus dispersal-mediated correlation. However, for the analyses in this report we make no assumptions about levels of dispersal between stocks. Instead, we examine the two possible extremes:

All stocks are completely connected with 100% dispersal. In this case, each stock i is merely a random weighted sample (weighted in the sense that some sample are larger than others) of the total ESU population. The m for the ESU level population can

be estimated by $\hat{m} = \text{mean}\left(\frac{\sum_i N_{i,t+t}}{\sum_i N_{i,t}}\right)$. The variance can be estimated using the same procedure as indicated in equation 10, with $\sum_i N_{i,t}$ in place of N_y :

$$\hat{s}^2 = \text{slope of var}\left(\frac{\sum_i \sum_j w_j S_{i,t+t-j}}{\sum_i \sum_j w_j S_{i,t-j}}\right) \text{ versus } t \quad [3.9]$$

The probabilities that the ESU level population is 90% declined at time t_e is calculated using [3.8] with \hat{m} and \hat{s} . This will overestimate extinction risk in general, since all stocks are assumed to have the same underlying population dynamics (i.e. rate of decline and variance).

All stocks are completely independent. In this case, the number in each stock i , N_i , fluctuate independently of the other stocks. The probability that the total ESU level population is 90% lower at time $t + t$ than at time t can be calculated as the probability that

$$\sum_i N_i(t+t) < x^* \sum_i N_i(t) \quad \text{where } N_i \sim N(\mathbf{m}_i t, \mathbf{s}_i t) \quad [3.10]$$

This will in general underestimate the risk of declines because index stocks with higher \mathbf{m} will not be affected by declines in stocks with lower \mathbf{m} .

Neither of the above assumptions is correct, but they represent the extremes between which the true value must lie. Results for the first assumption only (that all stocks are completely connected) are presented.

III.E.2. Data Used for ESU Level Analyses

For each ESU, all stock data that satisfied the following criteria were used:

- 1) Total live counts of natural adult spawners were available. In all cases, we attempted to use counts for wild spawners only. Counts for hatchery spawners, where available, were used in our calculations of hatchery effects.
- 2) Counts extended at least as far a 1995 (so that we have some representation of “current conditions”)
- 3) There were at least (*maximum age of return* + 4) counts available. This gives 4, 3 and 2 data points respectively for estimating $\text{var}(N_{t+i}/N_t)$ for $i = 1, 2$, and 3 respectively.

Data used to assess the potential impact of hatchery fish on population dynamics included either single-point estimates or time series of the percent hatchery fish in spawner counts. No information about reproductive rates of hatchery fish relative to wild fish was included at this time.

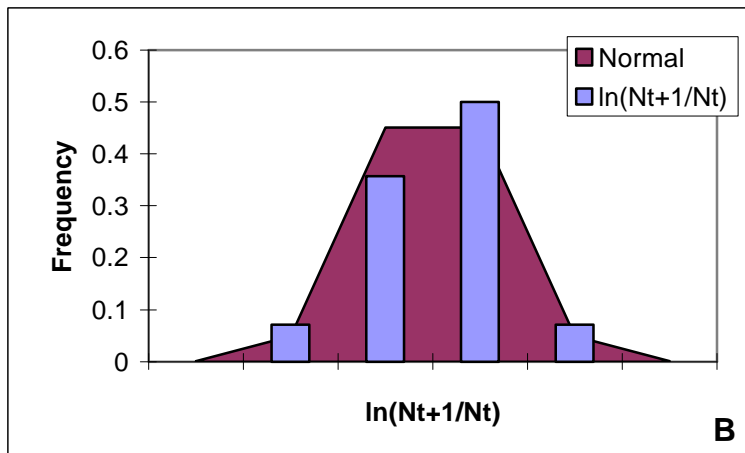
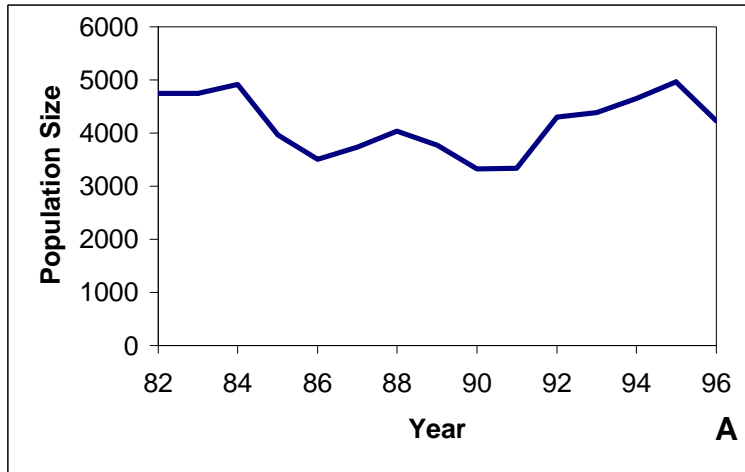


Figure III-1. Population trajectory and histogram of the natural log of population counts at time $t+1$ and t . These are data from Kalama River winter run steelhead. The population size is an estimate derived from a weighted sum of spawner counts as described below and in the appendix.

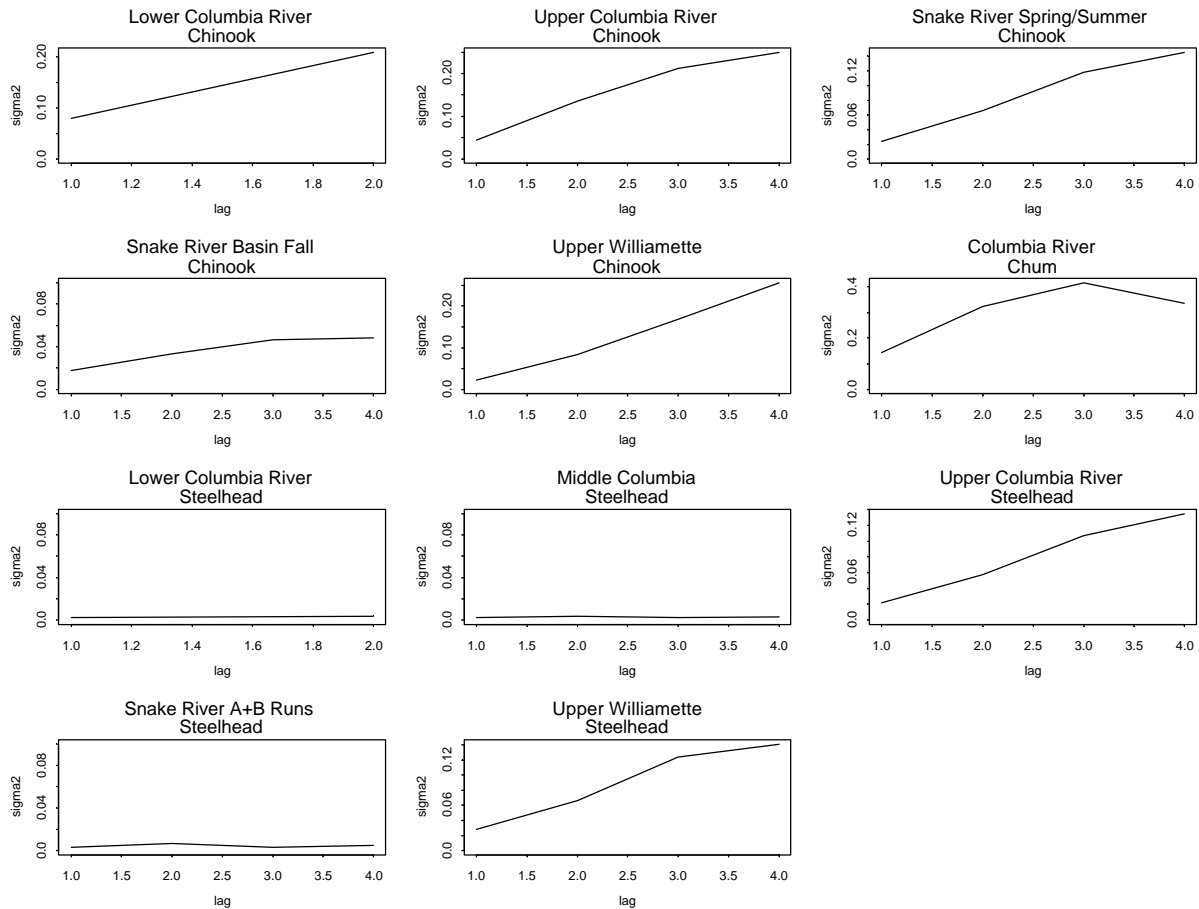


Figure III-2. The variance in $\ln(N_{t+4}/N_t)$ for $t = (1, 4)$ where N_t is the weighted sum of spawner counts as described in the text. A basic assumption of the analysis is that this relationship is linear. The slope of this relationship is used to estimate the variance of the total living fish that are current or future spawners (i.e., the population size at time t that we cannot observe). Plots that are flat indicate ESUs for which the variance was 0 or close to 0.

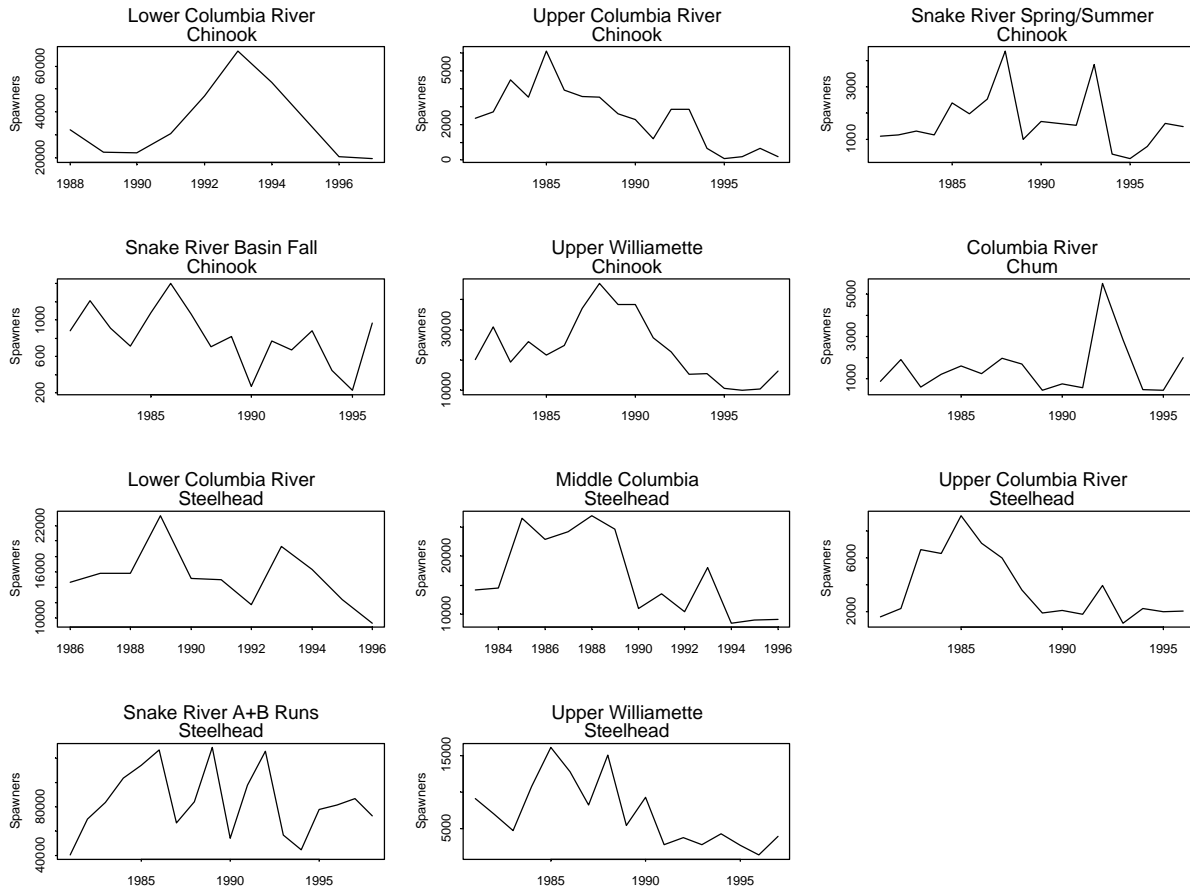


Figure III-3. Raw spawner counts used for the analyses. Counts include hatchery fish that spawn in the wild. The raw data is shown in the Appendix III.

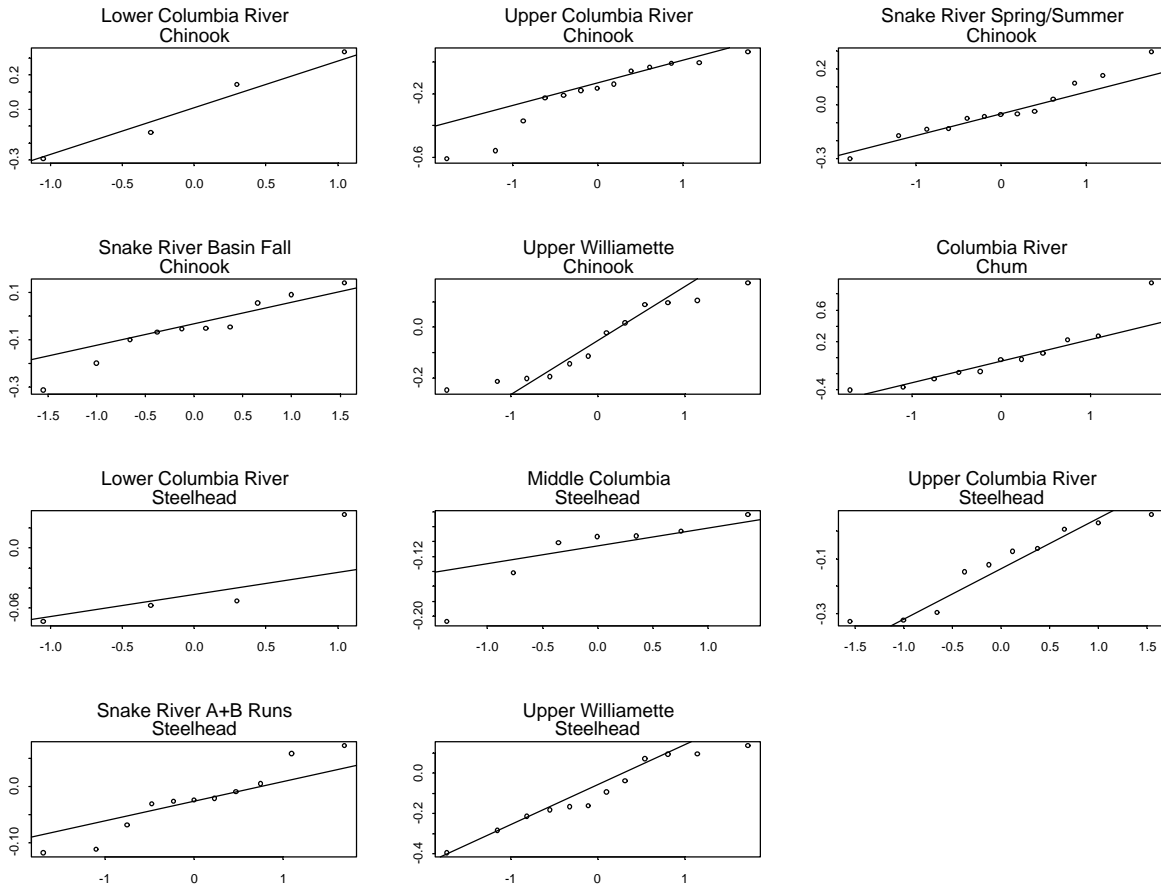


Figure III-4. Test of the normal distribution of $\ln(N_{t+1}/N_t)$ where N_t is the weighted running sum of spawners at time t . The line shows the expected values from a normal distribution. The data points should fall approximately on this line. While there are outliers, all except Snake River Steelhead are approximately linear. The x-axis is the quartiles of a standard normal and the y-axis is the $\ln(N_{t+1}/N_t)$ values.

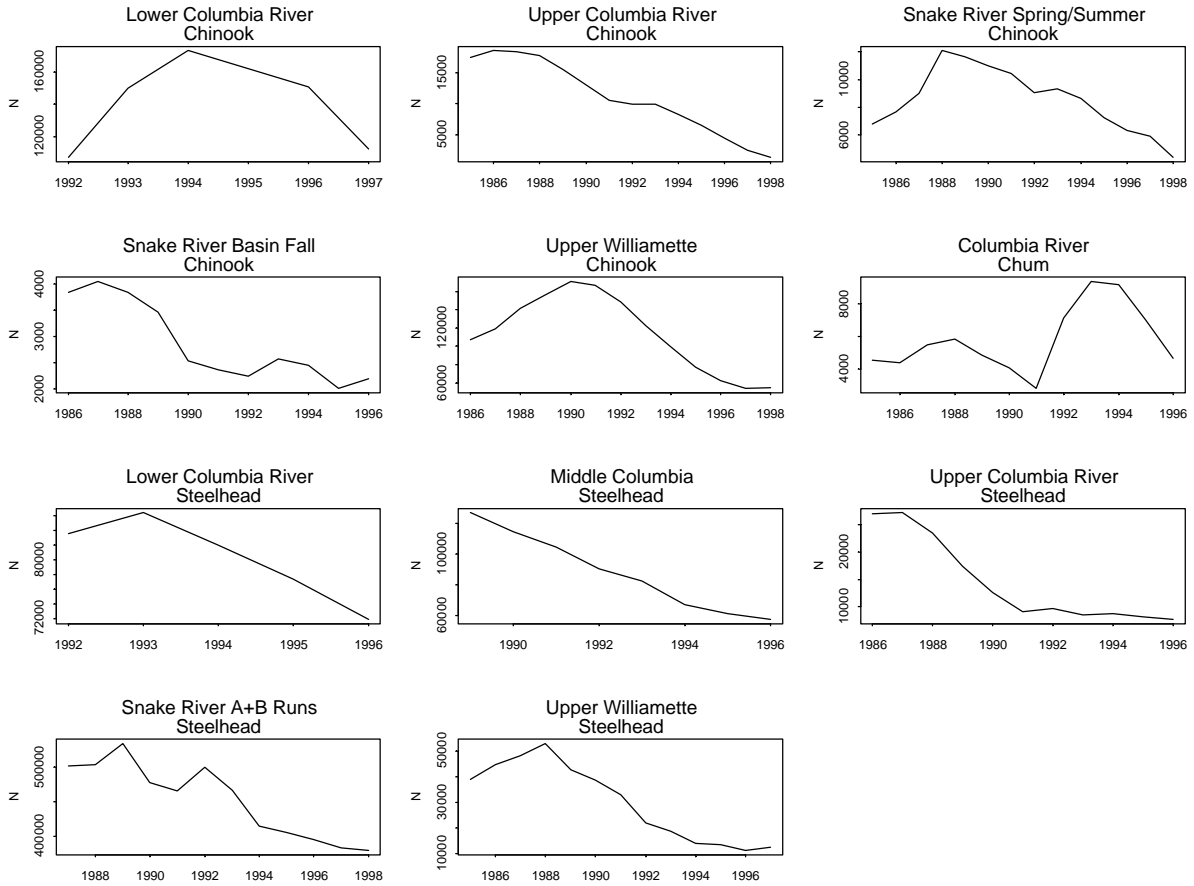


Figure III-5. Weighted running sums of spawner counts used in the analyses. As described in the text, this is an estimate of the living fish that are current spawners or that will survive to be future spawners (i.e., the total population size at time t which we cannot observe).

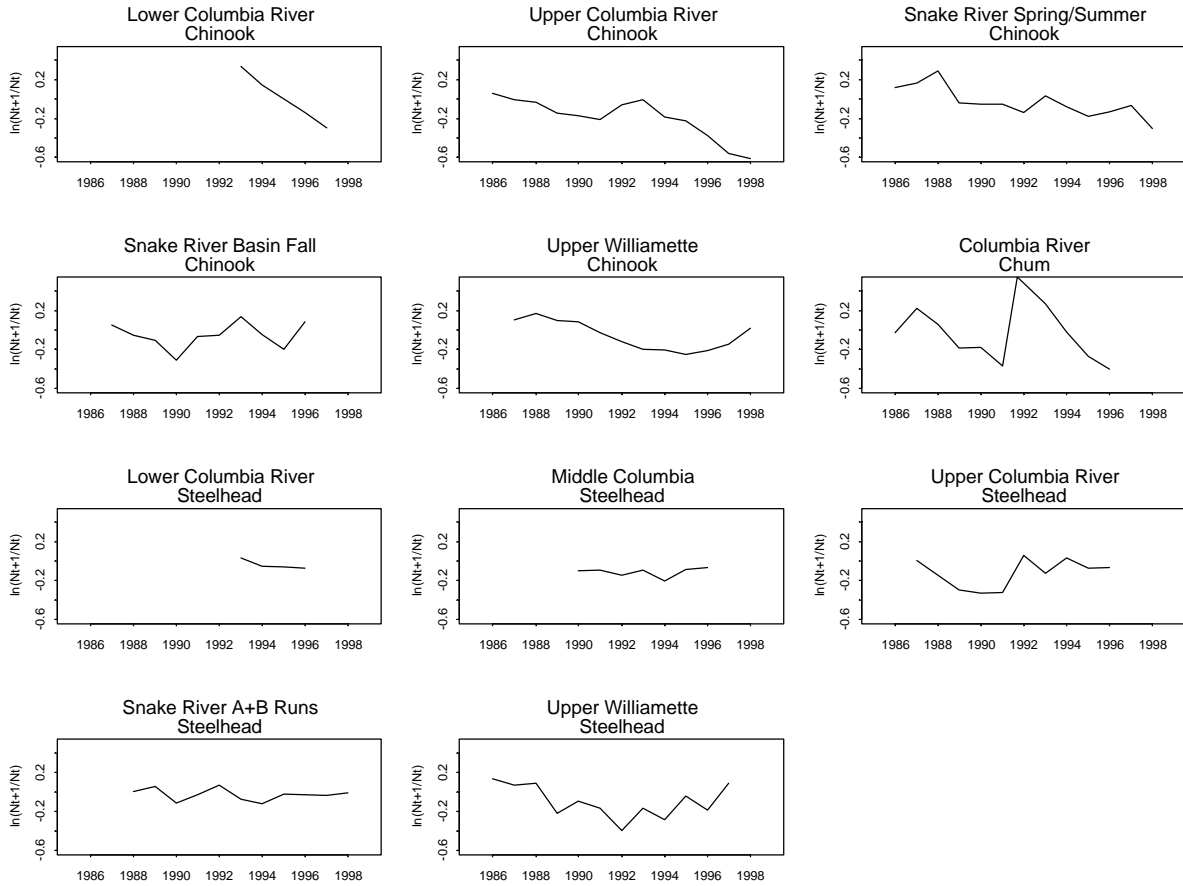


Figure III-6. $\ln(N_{t+1}/N_t)$ data which can be thought of as the $\ln(R/S)$ relationship. The analyses assume that there is no trend in this relationship. Two stocks show apparent trends in the 1980-present data used in these analyses: Upper Columbia River Chinook and Snake River Spring/Summer Chinook; however, an examination of longer time series (1970 to present) suggest that only the Upper Columbia River Chinook shows a downward trend (Figure III-7). Note that the Lower Columbia Chinook may show a trend but the data set is extremely short.

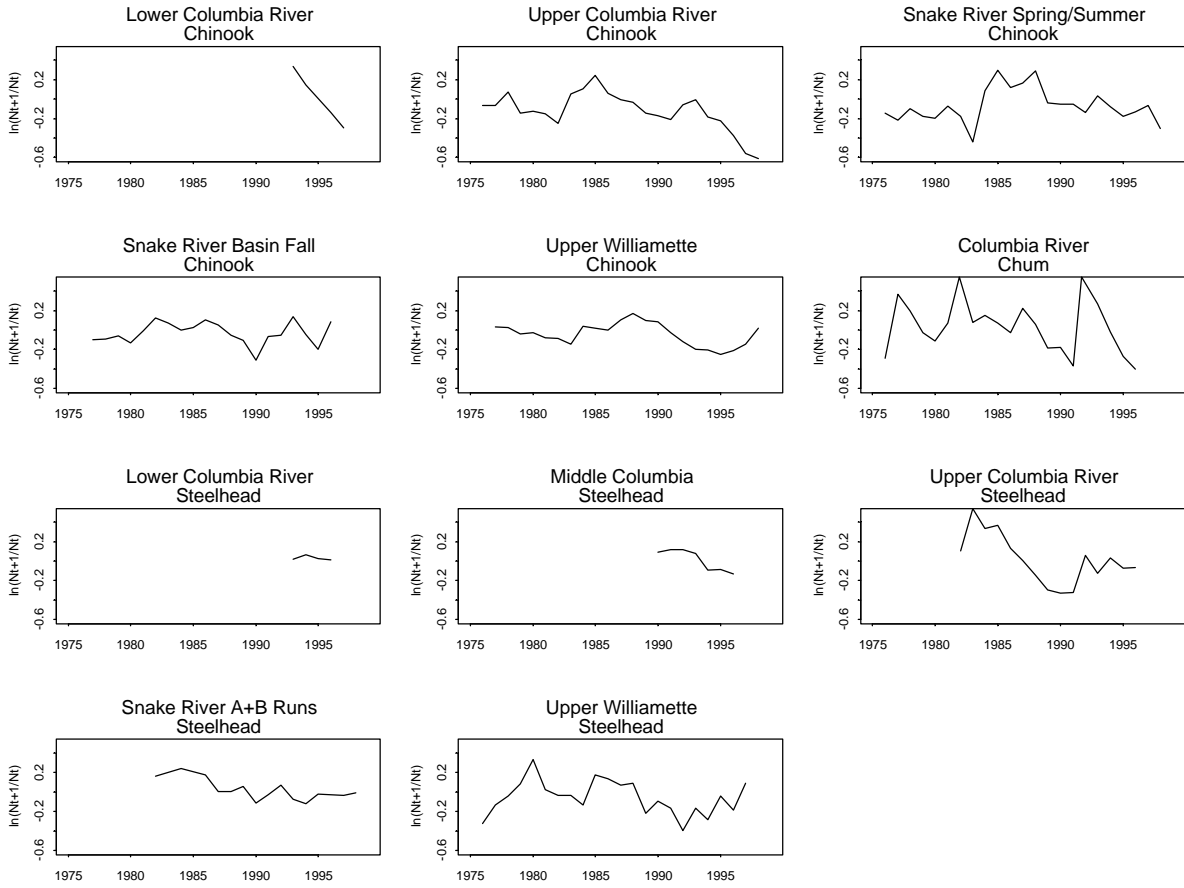


Figure III-7. $\ln(N_{t+1}/N_t)$ for 1970 to present where data is available.

Upper Columbia Spr. Ck. Population Growth
Rate

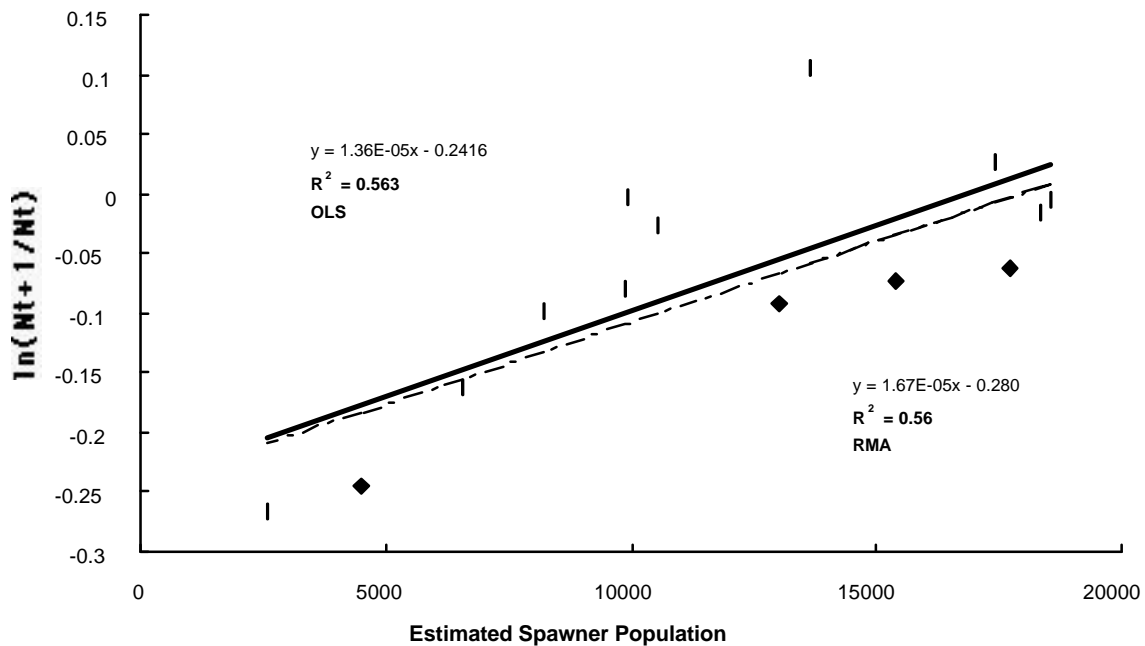
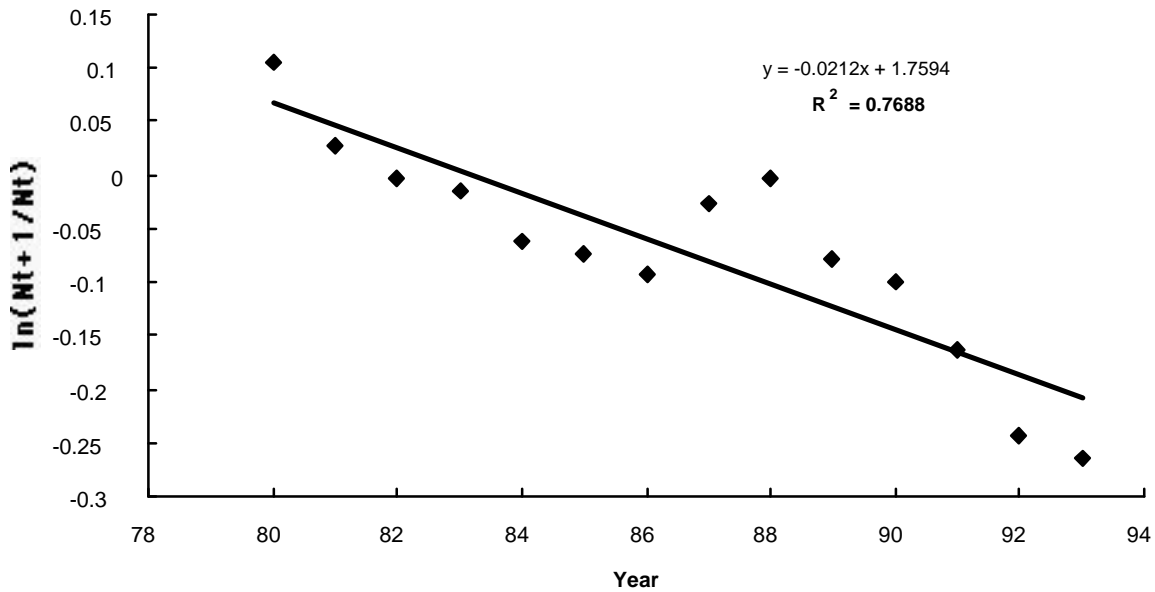


Figure. III-8. Trends in $\ln(N_{t+1}/N_t)$ for Upper Columbia Spring Chinook. Decreasing population growth rates in Upper Columbia Spring Chinook can be explained by either a trend in $\ln(R/S)$ (panel A), or by density depensation (panel B). The temporal trend in $\ln(R/S)$ explains more of the variance than its dependence on population size, but both regressions are significant at the $p < 0.05$ level.

IV. Standardized Comparisons of Risks across ESUs

IV.A. What is the current trend in spawner number and population size for individual ESUs?

Current trends in the number of spawners and the weighted running sum of spawner counts for individual stocks or ESUs are shown in Figures IV-1 and IV-2. Trends are strongly declining in most ESUs. Several ESUs exhibited peaks in spawner abundance in the early 1990's, including Lower Columbia Chinook, Upper Willamette Chinook and Columbia River chum.

IV.B. What is the rate of decline (λ) for individual stocks or ESUs?

Individual Stocks. For 68% of the individual stocks analyzed, λ was less than one (Figure IV-3). One third of these stocks had rates of decline less than 0.9, indicating these stocks are in severe decline. Population growth rates were increasing for the remaining 32% of individual stocks; the majority of positive λ values were less than 1.2.

ESUs. Among ESUs, λ was less than 1 for 9 of the 11 ESUs analyzed. Rate of population decline was less than 0.9 for four of these ESUs, indicating many ESUs are rapidly declining (Figure IV-4). The severity of decline is further indicated by the fact that upper confidence intervals were still less than 1. Population growth rate was slightly positive for lower Columbia Chinook and Columbia River Chum, although confidence intervals are especially large for both of these ESUs (due to high variance and short data sets) and include λ s less than 1.

Caveat. Note these estimates of λ do not account for any influence of hatchery fish. Accounting for the percentage of hatchery fish on the spawning grounds generally decreases these estimates of λ substantially (see analyses presented below).

IV.C. What is our best estimate of risk for individual stocks or ESUs?

IV.C.1. Methods for estimating risks to individual stocks or ESUs

IV.C.1.a. Comprehensive Risk Analysis

We completed a comprehensive risk evaluation that answered the following questions for stocks and/or ESUs:

(i) What is the risk of individual stocks reaching 1 fish/1 generation within the next 24/48/100 years? (note; this analysis is not possible for ESUs due to poor estimates of total population size within the ESU)

(ii) What is the probability of observing a 50% or 90% decline from current abundances within the next 24/48/100 years? (for stocks we show 90% and for ESUs we show 50% and 90%)

(iii) What is the most likely time for ESUs to reach their first decline to either 50% or 90% of current abundances?

Generally, all analyses were conducted for both ESU and stock-level data when possible. In some cases, however, data were not conducive to specific analyses. For example, estimating the risk of reaching 1 fish/1 generation was not feasible for ESUs because this analysis requires we know total abundance for the ESU. In most ESUs, data were available for only a subset of stocks. Thus, an estimate of total population size was not feasible. Because of the sheer number of analyses and subsequent results, we present a subset of the results in this section. Results of all combinations of stock/ESU, 24/48/100 years, 1 fish in one generation/50% decline/90% decline, with/without hatchery are presented in Appendix II.

IV.C.1.b. Role of Hatchery Fish

A critical uncertainty in estimating parameters describing population change in the modified Dennis analysis is the presence and influence of hatchery fish on the spawning grounds. Often the data neither indicate to what extent hatchery fish were excluded from the counts, nor whether there is any estimate of the fraction of spawners that are hatchery fish. Furthermore, even if the presence of hatchery fish is accurately documented, it is also important to know what their reproductive contribution might be (since their offspring will be counted as wild recruits). Because of lack of data on the relative reproductive output of hatchery born fish versus wild born fish, we present analyses under two extremes: naturally spawning hatchery fish produce no offspring and naturally spawning hatchery fish produce the same number of offspring as wild fish.

IV.C.2. Estimates of Risk

IV.C.2.a. Comprehensive Risk Analysis

(i) What is the risk of individual stocks reaching 1 fish/1 generation within the next 24/48/100 years? (note that this analysis is not possible for ESUs)

On average, the risk of individual stocks reaching 1 fish/1 generation is 12% (24 years) and 58% (100 years; Figures IV-5 and IV-6). Short-term risks of 12% are high; some chinook stocks have substantially greater risks. Overall, the risks were slightly higher for chinook compared to steelhead, resulting from higher variances in chinook stocks.

(ii) What is the probability of stocks/ESUs being at a 50% or 90% of current abundances at 24/100 years? The probability at 48 years is presented in tables in Appendix II.

The probability of individual stocks being at 90% of current abundances in 24 or 100 years is high. Among the 57 stocks analyzed, the average probabilities were 44% (24 years; median = 0.42) and 81% (100 years; median = 0.99). Results were comparable between chinook and steelhead stocks. Stock-level data are reported in Appendix II.

The probability that ESUs will be at 50% and 90% of current abundances in 24 or 100 years is extremely high (Figures VI-7 and VI-8). Among Chinook ESUs, probability of a 50% decline averaged 75% (24 years) and 86% (100 years). Probability of 50% decline among ALL steelhead ESUs was 100% in both the short and long term (24 and 100 years). ESUs with low probability of severe declines included Columbia River Chum and lower Columbia Chinook, although confidence intervals for population parameters of these two ESUs are extremely large (Figure VI-4).

Note these estimates of probability of decline do not account for any influence of hatchery fish. Accounting for the percentage of hatchery fish on the spawning grounds increases these probabilities (see analyses presented below).

(iii) What is the most likely time for ESUs to decline to either 50% or 90% of current abundances?

Lambda combines both m and s . The estimate of m , the instantaneous rate of population decline, represents one aspect of risk while s , the variability in m from year to year, represents another aspect. High variability is often associated with higher risk, however, its effects are not simply higher risk of reaching thresholds, such as extinction. Variability means not only the risk of a series of bad years but also a series of good years. Population trajectories from simulations that include variability in m (for example through variability in survivorships or fecundity) divide into two categories: trajectories that had a few good years and do much better than the mean behavior and trajectories that had a few bad years and hit lower thresholds quite quickly. In more precise terms, the distribution of population sizes at time t is lognormal.

Because the distribution of population sizes is strongly skewed, statistics such as the mean time to reach certain thresholds (such as a 50% decline) will be heavily influenced by a "long-tail" on the lognormal. Here we present the mode of the time to reach 50 and 90% declines. This represents the most likely time to hit these percent declines. It should be kept in mind that this will over-estimate the median time to reach these declines. The maximum likelihood estimate of the mode is (Dennis et al. 1991),

$$\log(x/1) / |m| \left\{ \left[1 + \frac{9s^4}{4(\log(x/1) |m|)^2} \right] - \frac{3s^2}{2 \log(x/1) |m|} \right\} \quad [4.1]$$

where $x = 2$ for 50% decline and 10 for 90% decline.

When looking at the results below, note that the most likely time to reach a threshold measures short-term risk while the probability that the stock is 50 or 90% lower than current levels in 24 or 100 years measures long-term risk. This is because this probability indicates where the stock will be in 24 or 100 years, but not what happened along the way. For example, a stock with $I > 1$ and high variance will increase and have a low probability that it is 50 or 90% below current levels in year 24 or 100, but may reach 50 or 90% declines (relative to current levels) in year 2 or 5, due to the high variability, but then increase well above these levels in subsequent years. The most likely (ML) time to first reach 50 or 90% declines (relative to current levels) captures this short-term risk. Because ML time captures short-term risk (i.e., is most strongly influenced by the variance σ^2) and the 24 and 100 year probabilities capture long-term risk (i.e. is most strongly influenced by m), these metrics can be very different. For example, chinook generally have higher variance than steelhead but lower m . Thus they tend to have higher short-term risk (lower ML time), but higher long term risk (24 and 100 year probabilities of 50 or 90% decline).

The estimates for each ESU are shown in Figure VI-9. Note that the most likely time to reach the threshold is only for those population trajectories that indeed reach the threshold. When m is positive, not all population trajectories reach the threshold and those that do, do so quickly. For this reason, the time to reach the threshold decreases as m increases (for positive m).

The most likely time until ESUs first decline to 50% of current abundance is extremely short, averaging 5.6 years across all ESUs. The ML time to decline was extremely short for chinook ESUs (average = 3 years) and slightly longer for steelhead ESUs (average = 8.7 years). These differences between chinook and steelhead are due to greater variance in chinook ESUs. The average time for ESUs to first decline 90% from current abundance was 25 years, with differences between chinook and steelhead ESUs similar to those observed at 50% declines. In both analyses, Columbia River Chum had a short time to reach 50% or 90% of current abundance. The Columbia River Chum ESU is consistently an outlier in our analyses, owing to its positive λ with large uncertainties.

IV.C.2.b. Role of Hatchery Fish

What effect does the presence of hatchery fish have on extinction risk for ESU/stocks?

A critical uncertainty in estimating the parameters describing population change is the presence and reproductive success of hatchery fish on the spawning grounds. Often the data do not make it clear to what extent hatchery fish were excluded from the counts, and whether there is any estimate of the fraction of spawners that are hatchery fish. Secondly, even if the presence of hatchery fish is accurately accounted for, it is also important to know what their reproductive contribution might be (since their offspring will be counted as wild recruits). We estimated population parameters in three ways: 1) without regarding hatchery influences (taking at “face value” that spawner

counts were wild fish); 2) with an estimate of fraction of spawners that were hatchery fish and assuming these hatchery born fish do not produce offspring; and 3) with an estimate of the fraction of fish that were hatchery fish and assuming that the hatchery fish reproduce at an equivalent rate to wild fish.

Given estimates of instantaneous rate of decline (m) and environmental variability (s^2), it is straightforward to calculate extinction risk metrics. If hatchery fish are not accounted for, but the fraction of spawners that they represent stays relatively constant from year to year, then m and s (and hence I) can still be effectively estimated (using the methods outlined in Section III). However, if the fraction of hatchery fish on spawning grounds varies widely over time, then information on that fraction through time and the relative fitness of those fish is essential to developing accurate descriptions of past population trends for wild fish.

We present results for the first and third approach to estimating population parameters (without regard to hatchery versus assuming a known fraction of hatchery fish reproduce at a rate equivalent to wild fish). Although we also did the analyses assuming that hatchery fish are present but do not reproduce, we do not present these results since this assumption will produce identical m and s estimates if the percent hatchery fish is relatively constant.

We present results for how I (for wild fish) changes when the proportion of hatchery fish is specified (s did not change appreciably and we do not show these results in the figures). Comparing estimates of the rate of population decline (I) from these two approaches allows us to begin to identify the extent to which hatchery fish can influence all of the estimates of risk presented above.

Estimates of percent hatchery fish were available for 41 stocks. Overall, the rate of population decline decreased from 0.95 to 0.62 (average lambdas across 41 stocks) when hatchery fish were included. This is a large change and would reflect a change of mean time (not ML time) to 90% decline from 46 years to 5 years.

Among ESUs the rate of population decline dramatically decreased when proportion of hatchery fish was specified (Figure VI-10). The average rate of population decline across ESUs, having accounted for hatchery influence when possible, is 0.58.

The implications of these results are tremendous for all estimates of risk assessment. In cases where available spawner counts include an unknown proportion of hatchery fish (i.e., we falsely assume all spawners are wild fish), estimates of risk are strongly optimistic (i.e., the estimated probabilities of extinction and decline are way too low). Knowing the percentage hatchery fish, if only at one point in time, allows us to begin to better examine risks for these ESUs. However, if no data are available to indicate how the percent hatchery fish has fluctuated over time, our abilities to evaluate risk for the wild fish is greatly diminished.

The calculations we present here assume that naturally spawning hatchery fish produce the same number of wild offspring as wild born fish. This is clearly an extreme assumption and the limited data available suggests that true relative reproductive output of hatchery fish to wild fish may be closer to 0.15 - *at least for some stocks*. However, our information on this critical parameter is extremely limited and without this information we can only present a most extreme scenario (hatchery fish produce as many offspring as wild fish).

However, under this most extreme scenario, we can see that steelhead wild populations are being heavily supported by releases of hatchery fish and production by these hatchery fish. Without hatchery releases (and without habitat or harvest changes to increase the in stream reproductive rates), the wild populations would be expected to decline to 10% of current levels within 2-4 years. Under the same, extreme 1 to 1 reproduction assumption, the wild reproducing Chinook populations were somewhat better. Expected time for wild populations without supplementation (and with no corresponding improvement to habitat or harvest levels) to decline to 10% of current levels would be 6-8 years. Under a less-extreme assumption regarding the reproductive output of hatchery fish, the masking effect of hatchery offspring would be diminished and the estimated lambda for wild fish would be closer to that presented in our analyses with no hatchery correction.

One of the obvious interpretations of this analysis is that hatcheries are supporting salmon stocks that are otherwise doomed for rapid extinction. However, a large body of research suggests that interbreeding between poorly adapted hatchery fish and wild fish and the consequent production of poorly adapted hybrids is one of the reasons that in-stream production is so low. Thus the low lambdas when hatchery percentages are high and hatchery fish are assumed to reproduce could also be viewed as evidence that hatchery interbreeding is severely compromising in-stream reproduction.

IV.D. How much improvement in lambda is needed to reduce risk?

One way that we can model the impacts that changes in habitat, hatcheries, harvest or hydropower have on salmon stocks is through modifying I , the rate of decrease (if I is less than 1) or increase (if I is greater than 1) for the population or stock. Obviously, opportunities for changing I will be different for different stocks, but as a first cut we evaluate how much improvement in I is needed to reduce estimates of risk.

We evaluated the degree of improvement necessary to reduce risk in two ways. First, we calculated the percent increase in I necessary to reduce the risk of extinction to less than 5% in 100 years. This is a standard metric used in risk assessment (IUCN 1994), making our analyses comparable with many others in the conservation literature. We also calculated the percent change in lambda necessary to reduce the probability of 90% decline in 100 years to less than 5%. These two analyses differ in the threshold to be avoided. In the first case, the threshold is absolute extinction (1 fish/1 generation); in the second case, the threshold to be avoided is a 90% decline from current

abundances. The second threshold is much less sensitive to errors in our estimate of total stock size or to the presence of non-reproducing hatchery fish in the counts. Both of these analyses were done at the stock level because this is the scale at which most management actions are likely to occur.

This analysis was done without including the effect that hatchery fish have on estimates of I . In situations where hatchery fish are present in the spawner counts *and* hatchery fish reproduce, the increases in I needed to minimize risk for the wild fish are far greater than those presented here.

For both analyses, the percent increase in I needed to avoid extinction or severe decline was variable among stocks (Figures VI-11 and VI-12). On average, a 10-15% increase is necessary, however the range increase spans 0-65%.

The percent increase in I necessary to prevent extinction or 90% decline depends on both m (the instantaneous rate of decline) and s^2 (the variability in that rate) – in a counterintuitive way. A stock can have a high probability of reaching a 90% decline in 100 years, but need a low percent increase in I if its variability in m is small. The individual chinook stocks that we analyze are small (individual creeks) with high variability while the individual steelhead stocks tend to be much larger (sometimes entire basins) with lower variability. For this reason, the percent increase in I required to prevent extinction or 90% decline is lower for steelhead than for chinook while the probabilities of reaching thresholds with no improvements is higher for steelhead than for chinook.

Having identified the amount of change necessary, we can then compare these changes with the potential for increases in I from specific actions such as changing harvest levels, improving habitat, reducing dam impacts, or reducing hatchery impacts.

IV.E. How likely are we to detect increases in lambda in 10 years?

The ESA mandates that threatened and endangered populations require recovery actions to mitigate the anthropogenic sources of risk. Recovery goals often involve increases in population size and reversing downward trends in population trajectories. In either case, to achieve recovery, population growth rates must be changed. Due to our poor understanding of the determinants of salmon population dynamics we may be forced to undertake management actions without a clear knowledge of their efficacy. Unfortunately, recovery actions may take 10s of generations to reach their goal, and as such, futile or even harmful actions may be in place for many years. Alternatively, successful actions that could be applied elsewhere may not exhibit their benefits rapidly, thus delaying their adoption into the salmon conservation repertoire. Can we use the population projection approach of the Dennis model to predict when a change in population growth rate is detectable? That is, given that a management action changes only the population growth rate, I , can we use our standard population projection

technique to examine the deviation of population trajectories with and without an altered growth rate?

In this manipulation we increase and decrease I by 1% and ask with what certainty can we detect a change in the population sizes resulting from the altered and unaltered trajectories in 10 years. To do this we project the populations forward using the standard Dennis approach,

$$\ln\left(\frac{N_{t+t}}{N_t}\right) = \mathbf{m}t + \mathcal{I}(\mathbf{m}t, \mathbf{s}^2t) \quad [4.2]$$

Where N_t is the population size at time t , t is the time interval over which the projection is made and $\mathcal{I}(\mathbf{m}t, \mathbf{s}^2t)$ is a normally distributed random variable of mean $\mathbf{m}t$ and variance \mathbf{s}^2t . To assess the likelihood of detecting a difference between the two stochastic population trajectories with and without a 1% increase in I , we calculate the separation of the resulting population size distributions assuming no sampling error. The log population sizes are normally distributed random variables $\mathcal{N}(\mathbf{m}t, \mathbf{s}^2t)$ and $\mathcal{N}(\mathbf{m}^+t, \mathbf{s}^2t)$, where $\mathbf{m}^+ = \mathbf{m} + \ln(1.01)$. Thus, the probability of detecting a change in log population size ($p < 0.05$) in 10 years due to a 1% change in I is the cumulative probability of the second distribution that lies to the right of $(\mathbf{m}t + 1.645\mathbf{s}t^{1/2})$.

For all Columbia River Basin stocks for which we have estimated \mathbf{m} and \mathbf{s}^2 we increased and decreased I by 1% and assessed the probability of detecting this change in 10 years (Figure III-13). It is sobering to note that in none of the stocks, would we have a > 50% chance of seeing the effect of an action in only 10 years. However, a 10% change I is much more likely to be detected and 28/53 stocks had a > 50% of showing an effect (Figure III-14). In general, it would be easier to detect an effect in steelhead populations than in chinook populations due to their more rapid rate of decline and lower variability in population size through time.

IV.F. Achieving needed improvements in lambda

One great challenge for salmon science is to link specific actions to hypothesized changes in lambda. Modifications in the arenas of harvest, habitat, hatcheries, hydropower and the higher ocean temperatures during El Nino years all have the potential to affect population growth rate. Unfortunately, few studies have specifically addressed the link between management actions and population growth rates at appropriate scales. Below (and in Section V), we present a first step toward assessing the potential to achieve needed changes in population growth rates through management actions. Much more thorough work is clearly needed not only in regards to the effect of management actions and the distribution of characteristics across the

landscape, but also with respect to ocean conditions, climate cycles and salmon productivity.

IV.F.1. Hydropower corridor

Modifications to the hydropower corridor that have the potential to affect population growth rate for Snake River chinook ESUs are discussed thoroughly in Section V. Discussion of hydropower modifications relevant to Upper Columbia ESUs will be provided in an update of this document.

In addition, we identify the number of dams present on the BPA GIS data-layer in each ESU in Table IV-1 (methods in Section IV.F.2.).

On average, there are about 27 dams present within ESU boundaries. To state the obvious, however, the potential to achieve improvements in population growth rate through modifications to dams and hydropower facilities vary tremendously from ESU to ESU, since the actual number of dams varies widely. In addition, dams not included within the ESU boundaries may still pose an obstacle during migration (both upstream and downstream) for some ESUs.

Table IV-1. Total area, and total number and density of dams and anadromous fish production facilities, in each of the 12 ESU's.

ESU Name	Area (km²)	Dams	Dams/1000 km²
<i>Lower Columbia River Chinook</i>	16,264.91	38	2.34
<i>SNAKE RIVER FALL CHINOOK</i>	35,531.93	14	0.39
<i>SNAKE RIVER SPRING/SUMMER CHINOOK</i>	58,158.62	30	0.52
<i>Upper Columbia River Spring Chinook</i>	18,146.36	14	0.77
<i>Upper Willamette River Chinook</i>	22,269.02	37	1.66
<i>Columbia River Chum</i>	10,988.11	17	1.55
<i>SNAKE RIVER SOCKEYE</i>	1,322.81	0	0.00
<i>Lower Columbia River Steelhead</i>	13,128.37	37	2.82
<i>Middle Columbia River Steelhead</i>	69,452.20	43	0.62
<i>SNAKE RIVER BASIN STEELHEAD</i>	76,060.28	34	0.45
<i>Upper Columbia River Steelhead</i>	24,748.43	43	1.74
<i>Upper Willamette River Steelhead</i>	12,654.84	19	1.50
Mean -->	29,893.82	27.17	1.20
Maximum -->	76,060.28	43.00	2.82
Minimum -->	1,322.81	0.00	0.00

IV.F.2. Habitats – Landscape level characteristics

IV.F.2.a. Land use characterization

Land use and land cover differ widely across the 12 ESUs. To broadly examine these areas, we characterized the general landscape within each of the 12 Columbia River Basin ESU's on the basis of dam and anadromous fish production facility density, and land use and land cover (LULC, Tables IV-1 - 5). We overlaid the boundaries for each ESU (NMFS 1999) with the three landscape geospatial datalayers using ESRI ARC/INFO. The geographic boundaries of individual ESU's only include freshwater areas where spawning and rearing occur. We characterized both the number and density of dams and production facilities (dams/production facilities/1000 km²) that occur within each ESU distribution. To examine the types of habitat within each ESU at a broad scale, we also quantified the percent of the total area in a range of land use and land cover habitat categories (34 categories total). We grouped the 34 second-level categories into broader land use or land cover units (Table IV-3).

Table IV-2. Summary table of geospatial data-layers used for landscape characterization for the 12 ESU's.

Datalayer	Source	Type	Scale	Comments
Land Use and Land Cover (LULC)	United States Geological Survey (USGS)	Polygon	1:250K	Land use and land cover generated using Anderson et al. (1976) protocols. Four or 16 ha minimum mapping unit (MMU). Late 1970's
Dams	Bonneville Power Administration (BPA)	Point	N/A	July, 1995
Anadromous Fish Production Facilities	Bonneville Power Administration (BPA)	Point	N/A	October, 1994

Table IV-3. Second level categories used in each general land use or land cover category.

		Category	Second Level Categories
L A N D U S E		Rangeland	Herbaceous Rangeland; Mixed Rangeland; Shrub and Brush Rangeland
		Cropland	Cropland and Pasture; Orchard, Grove, Vineyard, Nursery; Other Agricultural Land
		Urban	Mixed Urban or Built; Commercial and Services; Other Urban or Built-Up; Residential
		Industrial	Industrial; Industrial and Commercial Complexes; Transportation, Communications, Utilities
		Reservoirs	Reservoirs
		Strip Mines	Strip Mines
		Confined Feeding Operations	Confined Feeding Operations
		Category	Second Level Categories
L A N D C O V E R		Forest Land	Deciduous Forest Land; Evergreen Forest Land; Mixed Forest Land
		Alpine	Herbaceous Tundra; Shrub and Brush Tundra; Wet Tundra; Mixed Tundra; Glaciers; Perennial Snowfields
		Wetland	Forested Wetland; Non-Forested Wetland
		Bare	Bare Exposed Rock; Bare Ground; Beaches; Sandy Area (Non-Beach)
		Water Bodies	Lakes; Canals and Streams
		Transitional	Transitional Areas

Table IV-4. Land use characterization in each of the 12 ESU's.

ESU Name	Rangeland	Cropland	Urban	Industrial	Reservoir	Strip Mines	Confined Feeding Ops
Lower Columbia River Chinook	0.9393	7.6030	6.6295	0.8379	0.3559	0.1057	0.0087
Snake River Fall Chinook	22.8950	20.3402	0.2567	0.0351	0.3390	0.0208	0.0001
Snake River Spring/Summer Chinook	26.9259	10.3894	0.1868	0.0547	0.2251	0.0397	0.0005
Upper Columbia River Spring Chinook	21.2784	8.5429	0.5343	0.0686	0.4533	0.0403	0.0004
Upper Willamette River Chinook	0.4904	30.3431	4.5969	0.5025	0.2438	0.0920	0.0179
Columbia River Chum	1.4172	8.7389	8.2674	1.2161	0.5990	0.1333	0.0125
Snake River Sockeye	15.9286	2.3542	0.2178	0.1009	0.0700	0.1084	0.0000
Lower Columbia River Steelhead	0.9280	7.6736	7.4844	0.9391	0.4345	0.1257	0.0052
Middle Columbia River Steelhead	31.7046	25.9629	0.6706	0.2791	0.6612	0.0312	0.0042
Snake River Basin Steelhead	23.1428	11.9918	0.2015	0.0532	0.1755	0.0370	0.0004
U. Columbia River Steelhead	27.9963	18.8276	0.5191	0.2247	0.9527	0.0517	0.0035
U. Willamette River Steelhead	0.2012	39.8750	4.8634	0.4885	0.0688	0.0950	0.0283
Mean -->	14.4873	16.0535	2.8690	0.4000	0.3816	0.0734	0.0068
Maximum -->	31.7046	39.8750	8.2674	1.2161	0.9527	0.1333	0.0283
Minimum -->	0.2012	2.3542	0.1868	0.0351	0.0688	0.0208	0.0000

Table IV-5. Land cover characterization in each of the 12 ESU's.

ESU Name	Forest Land	Alpine	Wetland	Bare	Water Bodies	Transitional
Lower Columbia River Chinook Snake River Fall Chinook	80.1254	0.1545	0.6786	0.2862	2.2357	0.0385
Snake River Spring/Summer Chinook	54.9592	0.0975	0.0157	0.8642	0.1353	0.0180
Upper Columbia River Spring Chinook	59.7899	1.1615	0.0613	1.0262	0.0919	0.0184
Upper Willamette River Chinook	65.6111	1.8928	0.3916	0.6525	0.5130	0.0204
Columbia River Chum Snake River Sockeye	62.9377	0.1063	0.3482	0.0299	0.2003	0.0800
Lower Columbia River Steelhead	75.0198	0.0434	0.9949	0.2521	3.2621	0.0421
Middle Columbia River Steelhead Snake River Basin Steelhead	65.2998	8.6963	0.6905	5.2012	1.2797	0.0054
U. Columbia River Steelhead	80.0688	0.1914	0.4189	0.2965	1.3996	0.0332
Upper Willamette River Steelhead	40.3472	0.0570	0.1647	0.0098	0.0593	0.0411
Mean -->	62.1189	0.9033	0.0513	1.1715	0.1139	0.0157
Maximum -->	48.6822	1.3878	0.2986	0.4882	0.4163	0.1508
Minimum -->	53.6682	0.0000	0.3728	0.0018	0.2127	0.1184
	62.3857	1.2243	0.3739	0.8567	0.8266	0.0485
	80.1254	8.6963	0.9949	5.2012	3.2621	0.1508
	40.3472	0.0000	0.0157	0.0018	0.0593	0.0054

Some ESUs had high percentages of rangeland (Upper Columbia and Snake), cropland (upper Willamette, urbanization (Lower Columbia, Upper Willamette), and forest cover (Lower Columbia; Tables IV-4 - 5). Although percentages of area attributable to industry, reservoirs, strip mines, feeding operations, bare cover and wetlands are small, land uses/covers can have notable impacts on freshwater habitats.

The landscape characterization is intended only as a qualitative reference. The characterization cannot be used for quantitative spatial analyses because: the spatial scale of the geospatial datalayers is too coarse; the characterization is based on static

data; and the LULC datalayer is too old. The scale over which Pacific salmon population dynamics and biology occurs, ranges from the reach (a few tens of meters) all the way up to the sub-basin (thousands of square kilometers). In addition, habitat/fish interactions vary considerably with species, stock, and sub-basin biogeography. The complexity of these interactions is not captured in our characterization. Finally, some of the ESU's spatially overlap with each other, violating the assumption of independence required for doing statistically rigorous hypothesis testing.

IV.F.2.b. Preliminary studies linking habitat characteristics and productivity

A study by Regetz (in review) provides some evidence for opportunities to improve productivity through changes in freshwater habitat characteristics. Regetz investigated correlations between recruits per spawner of 22 Columbia basin stocks to the following 10 landscape-scale habitat variables: (1) urban/built land cover, (2) agriculture/rangeland land cover, (3) forested land cover, (4) distribution of grazing allotments, (5) the number of mining operations per square kilometer of total subwatershed area, (6) the summed length of EPA 303(d)-listed streams per total streamlength within the subwatersheds ("EPA 303(d)" refers to a section of the Federal Water Pollution Control Act requiring states to list all waterbodies for which current pollution controls are insufficient to meet water quality standards), (7) master watershed sensitivity index (MWSI), (8) riparian integrity, (9) disturbance recovery potential, and (10) erosion potential. Three of these environmental variables: percent of land classified as urban or "built-up," proportion of stream length failing to meet water quality standards, and an index of the ability of streams to recover from sediment flow events accounted for over 60% of the variation in mean salmon recruitment. Interestingly, within the areas corresponding to the 22 index stocks, the percent of urban/built land cover was quite small (<1%) even in the "worst" locations suggesting that stocks are sensitive to even minor variations in urban development.

Furthermore, it appears that these landscape attributes may limit the maximum recruitment rates of salmon, with a magnitude of differences in productivity large enough to be relevant to recovery planners. For example, Regetz reports that if the worst index stocks with respect to 303(d) listings could be improved to water quality levels equivalent to the best stocks, then, in the median case, the maximum number of recruits per spawner would nearly double. Plots of arithmetic mean and maximum R/S against numerous habitat attributes often share a similar "wedge" pattern, with most or all data points falling below a diagonal line from the upper left to the lower right (Figure IV-15). Productivity can clearly be low even when habitat quality (measured at the landscape level) seems to be very high. However, in all cases the most productive populations occur only where landscape-level habitat conditions are good, whereas index areas characterized by low quality habitat are almost invariably associated with lowered stock productivity. This pattern suggests that some sub-watershed scale habitat attributes might act as limiting factors, whether or not they are functionally related to population performance. In other words, although good habitat conditions do not necessarily correspond to high salmon productivity (because myriad other factors

can still depress populations), diminishing habitat quality leads to lower salmon productivity.

IV.F.2.c. General

Clearly, the degree to which actual post-management recruitment rates actually match any predicted increase will depend on the relative importance of non-habitat factors. In addition, the potential for achieving improvements, and the kinds of changes that can be made will vary substantially from ESU to ESU. Even so, it is apparent that there is much scope for habitat restoration that can be translated into significant demographic improvement.

IV.F.3. Hatcheries – Distribution and preliminary analyses

IV.F.3.a. Distribution of Hatcheries within ESUs

We also summarized the number and density of anadromous fish production facilities within ESU boundaries (Table IV-6).

While the average ESU has approximately 15 production facilities within its boundaries, again there will be widely varying opportunities to affect population growth rate through hatchery actions. Potential ecological and genetic effects of hatchery fish on wild populations have been well described (NMFS 1999). However, the rates and occurrence of these effects has not been well quantified. Therefore, the potential to achieve changes will depend on: where effects occur (e.g., freshwater habitat or estuary), the mechanisms by which they occur (e.g., predation or competition for ecological effects), and the rates at which they occur (e.g., the rate at which hatchery and wild fish breed, and the reproductive success of hatchery fish).

In total, nearly 100 facilities in the Columbia River Basin release approximately 150 million smolts annually. Production by hatcheries is presently capped and hatchery operation goals and principles are undergoing a congressionally-mandated review.

IV.F.3.b. Preliminary analyses

The use of hatcheries to augment harvest is based on the assumptions that (1) the freshwater environment limits natural production; (2) carrying capacity of the ocean exceeds the production of natural populations; and most importantly, (3) hatchery production will not negatively impact natural populations. The degree to which production hatcheries impact population growth rates of natural fish is largely unknown. However, a preliminary, unpublished analysis of smolt to adult returns from 1975 to the present suggests that hatchery production can dramatically impact survival of wild fish (Figure IV-16). During years of poor ocean conditions (defined here as positive values of the El Niño – Southern Oscillation Index) survival rates of wild fish varied inversely with the number of hatchery fish entering the ocean with a four-fold greater survival when hatchery releases were lowest compared to when they were highest. In contrast,

this effect was absent when ocean conditions were better (negative El Niño – Southern Oscillation Index values). The interaction of ocean conditions with hatchery releases allows the potential for management to modify hatchery production during periods likely to induce higher ocean mortality. This work is preliminary, but it indicates at a minimum that additional work in this vein will help to identify areas where improvements in population growth rate might be achieved through modification of hatchery management practices.

Table IV-6. Total area, and total number and density of anadromous fish production facilities, in each of the 12 ESU's.

ESU Name	Area (km²)	Production Facilities	PF/1000 km²
Lower Columbia River Chinook	16,264.91	35	2.15
Snake River Fall Chinook	35,531.93	12	0.34
Snake River Spring/Summer Chinook	58,158.62	15	0.26
Upper Columbia River Spring Chinook	18,146.36	11	0.61
Upper Willamette River Chinook	22,269.02	9	0.40
Columbia River Chum Snake River Sockeye	10,988.11	33	3.00
Lower Columbia River Steelhead	1,322.81	1	0.76
Lower Columbia River Steelhead	13,128.37	23	1.75
Middle Columbia River Steelhead	69,452.20	9	0.13
Snake River Basin Steelhead	76,060.28	20	0.26
Upper Columbia River Steelhead	24,748.43	13	0.53
Upper Willamette River Steelhead	12,654.84	4	0.32
Mean -->	29,893.82	15.42	0.88
Maximum -->	76,060.28	35.00	3.00
Minimum -->	1,322.81	1.00	0.13

IV.F.2. Sensitivity of Annual Population Growth Rate to Changes in Harvest

We determined total (ocean and in-river) exploitation rate for each ESU using estimates from the Pacific Salmon Commission, ODFW, WDFW, and IDFG, as well as PATH run-reconstructions for Snake River, and the QAR process for the Upper Columbia River. Because the bulk of the spawner counts we used were restricted to the 1980s and early 1990s, we used average harvest rates during this time period. However, in the mid to late 1990s, ESU concerns resulted in the drastic reduction of harvest rates.

The absolute value of lambda with varying rates of harvest was determined by:

$$\lambda_{\text{altered}} = \lambda_{\text{observed}} \left(\frac{1}{1-h+d} \right)^{1/\text{generation time}} \quad [4.3]$$

where h is the total exploitation rate and d is the target harvest rate.

Because both analyzed (i.e., 1980 - early 1990s) harvest rates and generation time influence the altered harvest rate, ESUs vary in their response to changes in those rates. Thus, harvest reductions or moratoria will have the greatest potential for increasing population growth rates in those ESUs with short generation times and that experience high harvest rates (Figure IV-17).

If harvest is eliminated entirely, increases from 20 to 30% in lambda can be achieved in Lower Columbia Chinook, Upper Willamette Chinook, and Snake River Fall Chinook (Figure IV-17, Table IV-8). In these ESUs rates of population growth increase from 1.085 to 1.412, 1.136 to 1.437 and 0.951 to 1.168, respectively (Figure IV-17, Table IV-7, 8). In contrast, for those ESUs that experience low harvest rates, the maximum possible increase in average annual population growth rate that can be achieved through harvest management is 1-2 percent (Figure IV-18, Table IV-7, 8). All Steelhead ESUs have very low population growth rates. As a result, even when harvest rates are high, total elimination of steelhead harvest results in lambdas less than, or only slightly greater than 1.00, even though lambda would show an increase. As mentioned before, harvest rates in some cases have already been reduced due to ESA concerns. Note also that we do not include Columbia River Chum salmon in these analyses since estimates of the total population size (and therefore harvest rates) are not currently available.

TableIV-7. Predicted annual population growth rate with given harvest r

	Harvest Rate			Adjusted Annual Population Growth Rate											
	λ	1980s to early 1990s	Mean Return Time	55% Harvest	50% Harvest	45% Harvest	40% Harvest	35% Harvest	30% Harvest	25% Harvest	20% Harvest	15% Harvest	10% Harvest	5% Harvest	0% Harvest
Chinook															
Lower Columbia	1.074	0.58	3.29	1.084	1.101	1.120	1.140	1.162	1.186	1.213	1.241	1.274	1.310	1.350	1.397
U. Columbia Spring	0.878	0.09	4.25	0.803	0.810	0.817	0.824	0.831	0.839	0.848	0.857	0.866	0.876	0.886	0.898
Snake R. Spr/Sum	0.981	0.06	4.73	0.901	0.908	0.915	0.922	0.929	0.937	0.945	0.954	0.963	0.973	0.983	0.994
Snake R. Fall	0.931	0.53	3.67	0.926	0.939	0.953	0.967	0.983	1.000	1.019	1.039	1.061	1.086	1.113	1.144
Upper. Willamette	0.949	0.65	4.47	0.972	0.984	0.997	1.012	1.028	1.045	1.064	1.085	1.108	1.134	1.165	1.200
Steelhead															
L. Columbia Winter	0.939	0.31	4.47	0.895	0.903	0.912	0.921	0.931	0.941	0.952	0.964	0.977	0.990	1.005	1.020
L. Columbia Summer S	0.956	0.04	5.17	0.883	0.889	0.895	0.901	0.908	0.914	0.922	0.929	0.937	0.945	0.954	0.964
Mid Columbia	0.882	0.28	5.17	0.842	0.849	0.855	0.863	0.870	0.878	0.887	0.896	0.906	0.916	0.928	0.940
Upper. Columbia	0.873	0.35	3.78	0.832	0.841	0.851	0.862	0.873	0.885	0.897	0.911	0.926	0.942	0.959	0.978
Snake R. Aggregate	0.965	0.2	5.17	0.911	0.918	0.925	0.932	0.940	0.948	0.956	0.965	0.975	0.985	0.996	1.008
Upper Willamette	0.872	0.21	4.08	0.812	0.819	0.827	0.836	0.845	0.854	0.864	0.874	0.886	0.898	0.910	0.924

Table IV-8. Percent change in lambda with given harvest

	λ	Harvest Rate		Percent change in Average Annual Growth Rate											
		1980s to early 1990s	Mean Return Time	55% Harvest	50% Harvest	45% Harvest	40% Harvest	35% Harvest	30% Harvest	25% Harvest	20% Harvest	15% Harvest	10% Harvest	5% Harvest	0% Harvest
Chinook															
L. Columbia	1.074	0.58	3.29	0.930	2.567	4.324	6.218	8.268	10.500	12.944	15.639	18.632	21.989	25.796	30.171
U. Columbia Spr	0.878	0.09	4.25	-8.519	-7.766	-6.979	-6.156	-5.293	-4.386	-3.432	-2.426	-1.362	-0.234	0.965	2.244
Snake R. Spr/Sum	0.981	0.06	4.73	-8.086	-7.420	-6.726	-6.001	-5.242	-4.446	-3.611	-2.732	-1.806	-0.826	0.213	1.317
Snake R. Fall	0.931	0.53	3.67	-0.539	0.834	2.301	3.872	5.563	7.391	9.376	11.545	13.930	16.574	19.531	22.873
U. Willamette	0.949	0.65	4.47	2.384	3.701	5.117	6.645	8.303	10.112	12.101	14.304	16.766	19.550	22.740	26.461
Steelhead															
L. Columbia Winter	0.939	0.31	4.47	-4.699	-3.818	-2.889	-1.910	-0.874	0.225	1.394	2.642	3.978	5.416	6.970	8.658
L. Columbia Summer	0.956	0.04	5.17	-7.663	-7.060	-6.431	-5.775	-5.090	-4.373	-3.621	-2.831	-1.999	-1.121	-0.192	0.793
Mid Columbia	0.882	0.28	5.17	-4.519	-3.774	-2.992	-2.169	-1.300	-0.382	0.591	1.626	2.731	3.914	5.186	6.562
U. Columbia	0.873	0.35	3.78	-4.704	-3.626	-2.487	-1.281	0.000	1.365	2.823	4.388	6.074	7.899	9.884	12.058
Snake R. Aggregate	0.965	0.2	5.17	-5.642	-4.950	-4.226	-3.466	-2.668	-1.827	-0.940	0.000	0.997	2.060	3.195	4.412
U. Willamette	0.872	0.21	4.08	-6.922	-6.050	-5.136	-4.174	-3.160	-2.090	-0.957	0.247	1.528	2.897	4.366	5.948

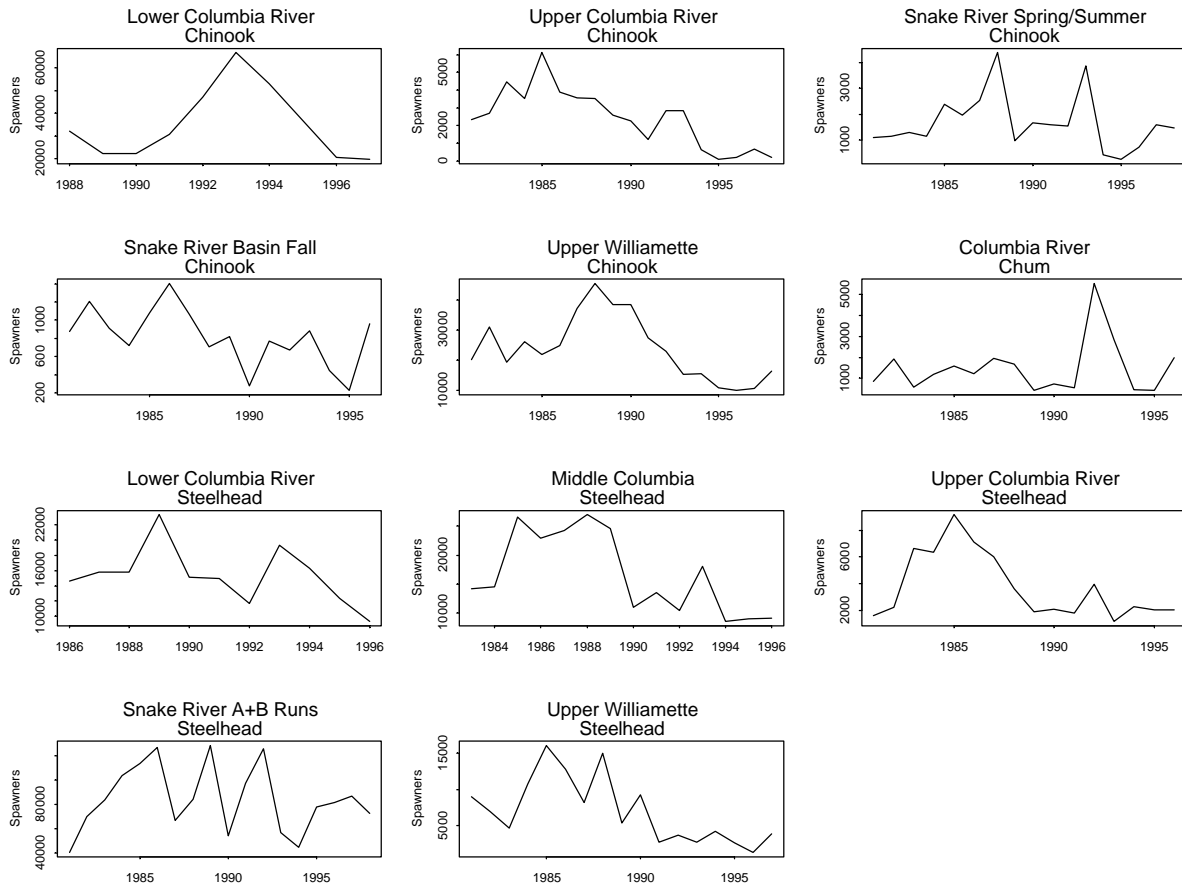


Figure IV-1. Raw spawner counts used for the analyses. Counts include hatchery fish that spawn in the wild.

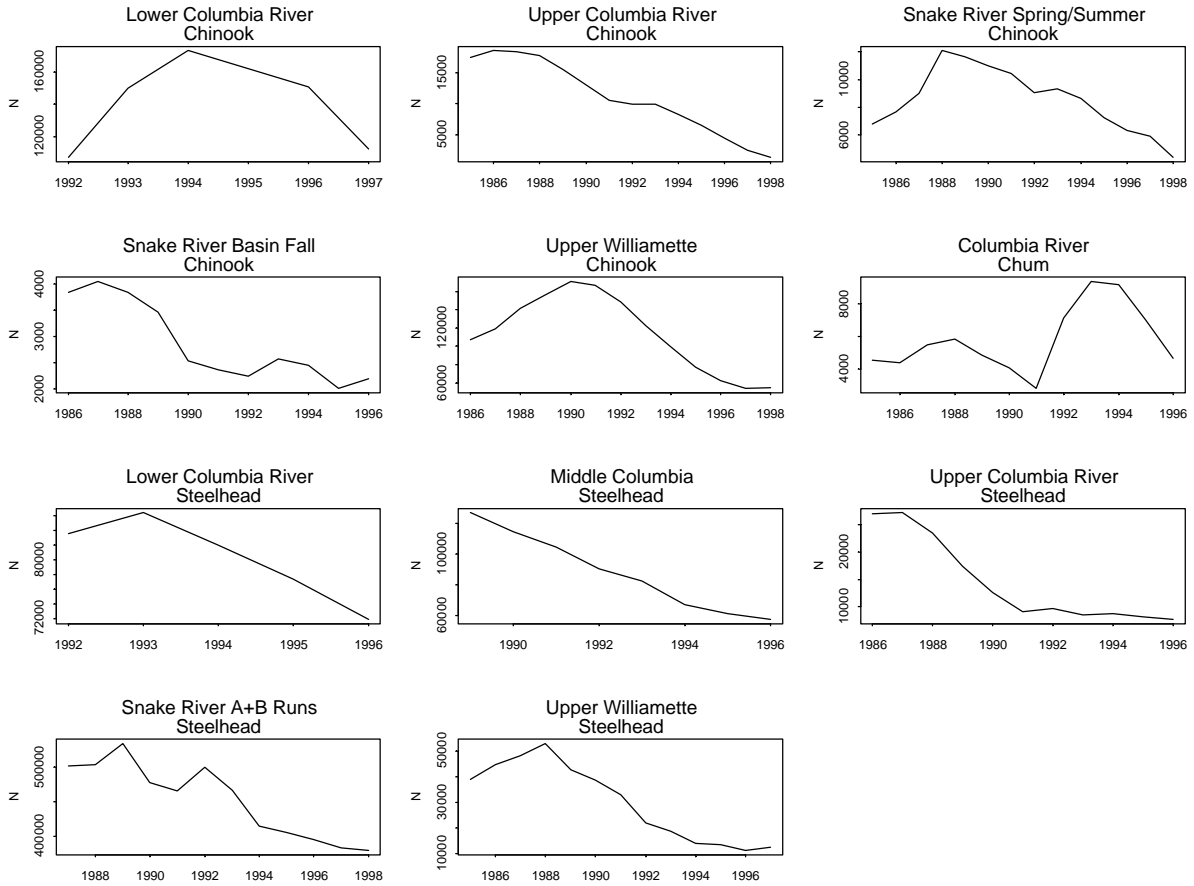


Figure IV-2. Weighted running sums of spawner counts used in the analyses. As described in the text, this is an estimate of the living fish that are current spawners or that will survive to be future spawners (i.e., the total population size at time t which we cannot observe).

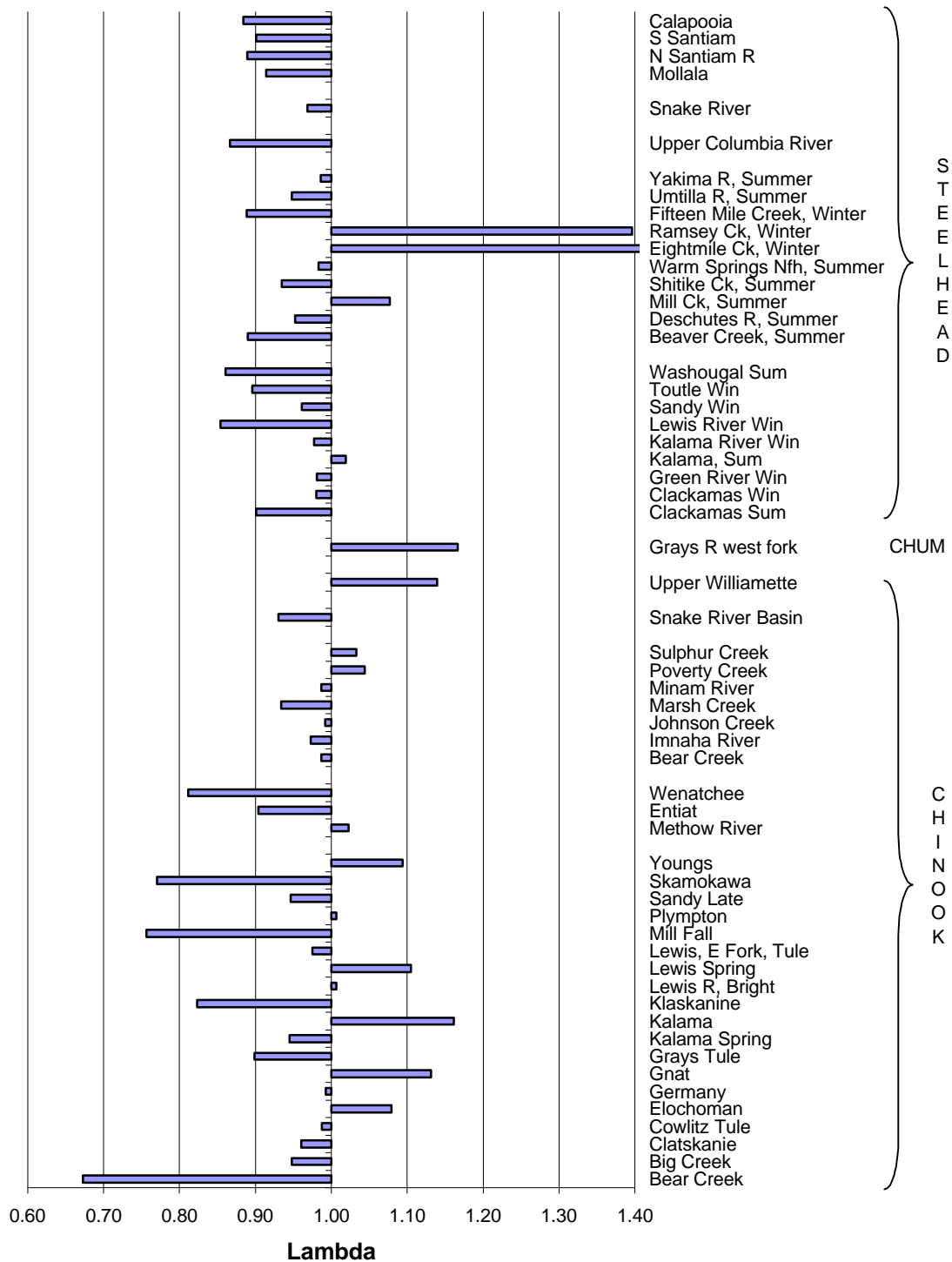


Figure IV-3. Estimated lambda at the stock level.

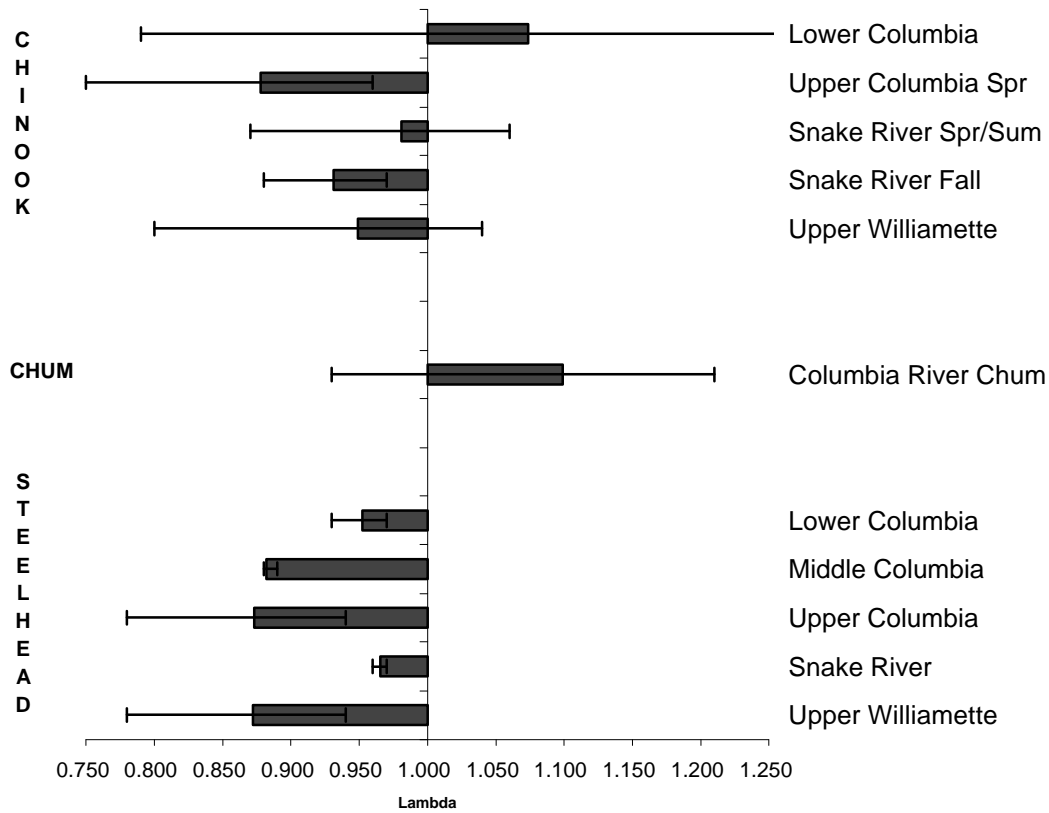


Figure IV-4. Estimated lambda at the ESU level. The error bars indicated the confidence that the estimated lambda reflects the long-term rate of decline (or increase) rather than a short-term trend. ESUs with large error bars indicate ESUs with high variability and short data sets. Note that the error bars do not reflect uncertainty due to sampling error – and thus our estimate of the true short-term rate of decline.

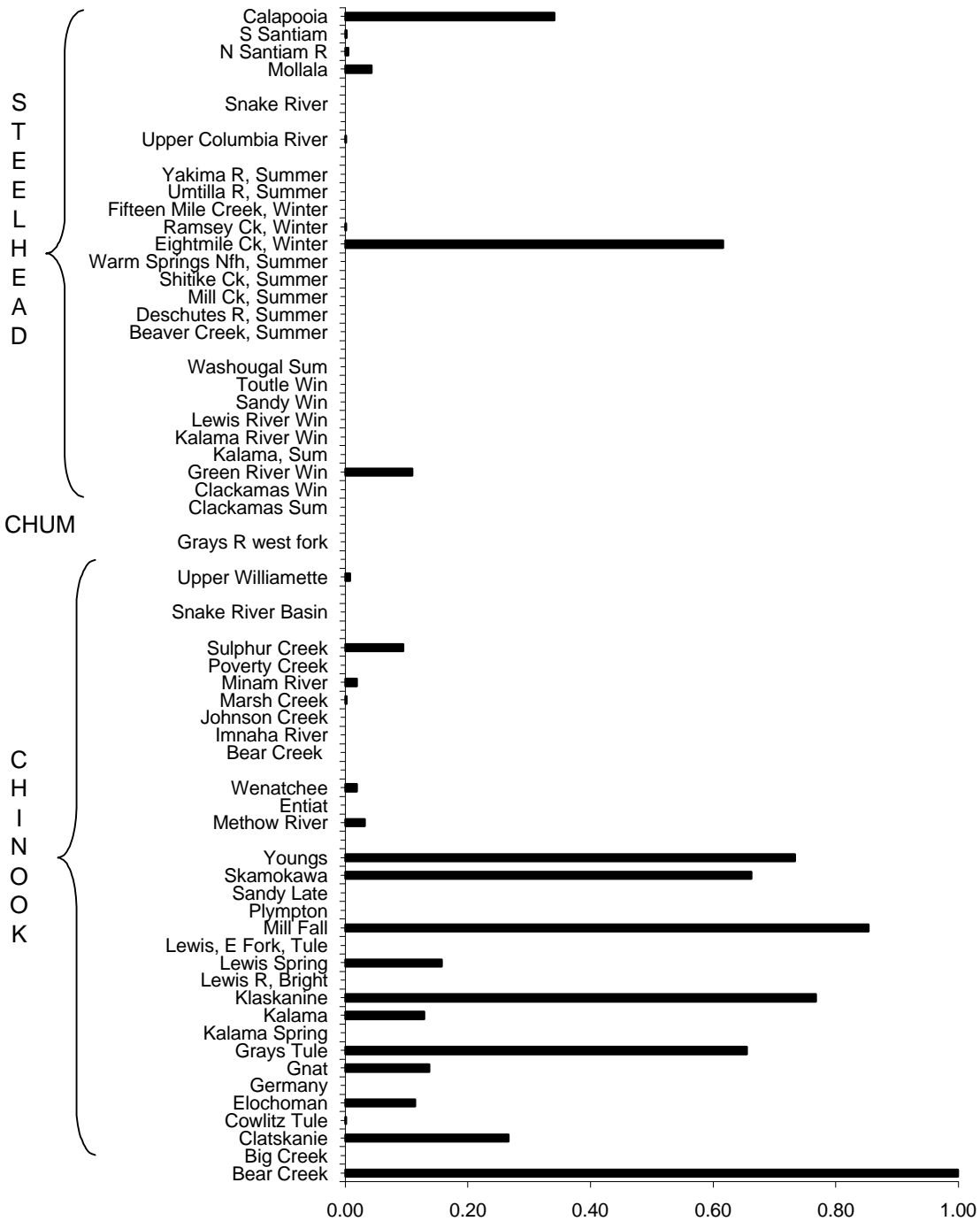


Figure IV-5. Probability of extinction at the stock-level in 24 years. Extinction is defined as one fish in one generation.

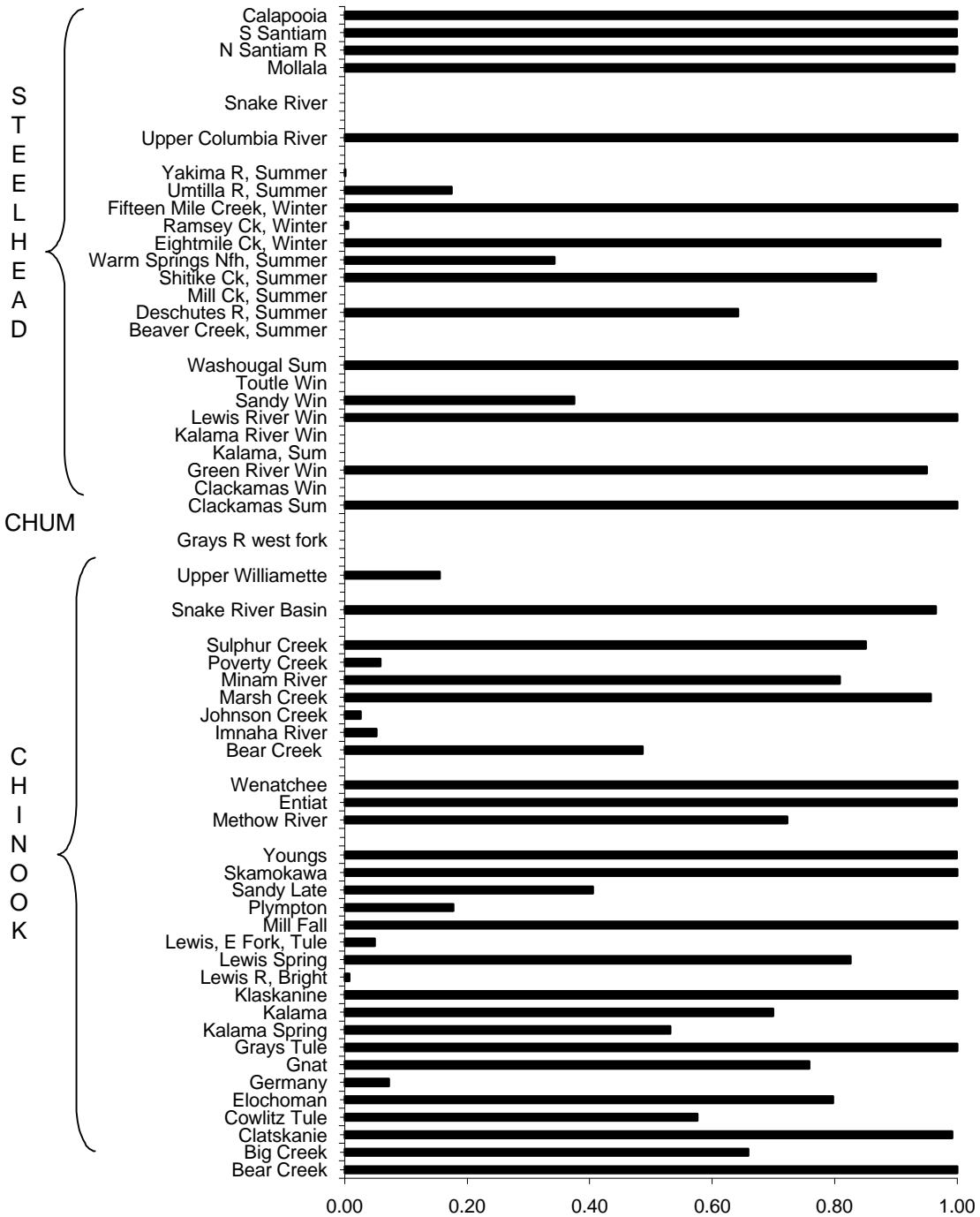


Figure IV-6. Probability of extinction at the stock-level in 100 years. Extinction is defined as one fish in one generation.

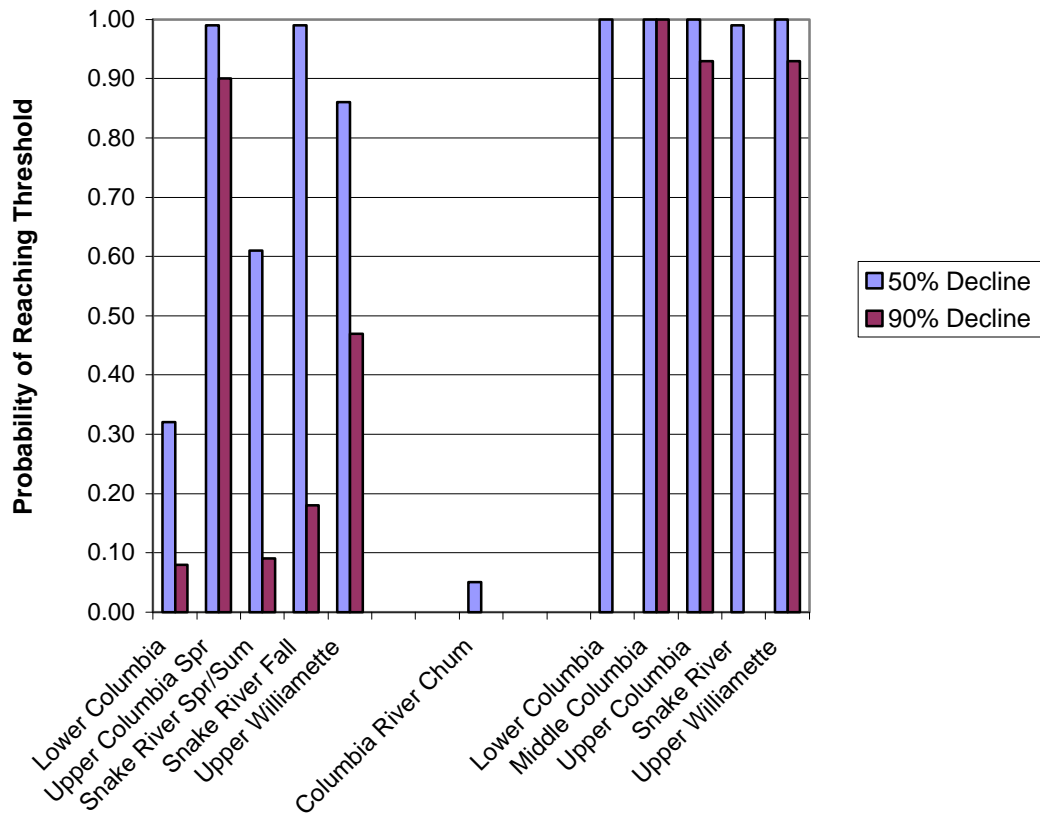


Figure IV-7. Probability that ESU is 50 or 90% below current levels at 24 years in the future. Note that this is not the probability that the ESU has dipped 50% or 90% below current levels within 24 years. This probability is better reflected by the most likely time to first hit a 50 or 90% decline. Note that ESUs with a positive lambda may have a short most likely time to first hit a 50 or 90% decline but a small probability that the stock is still 50 or 90% declined in 24 years.

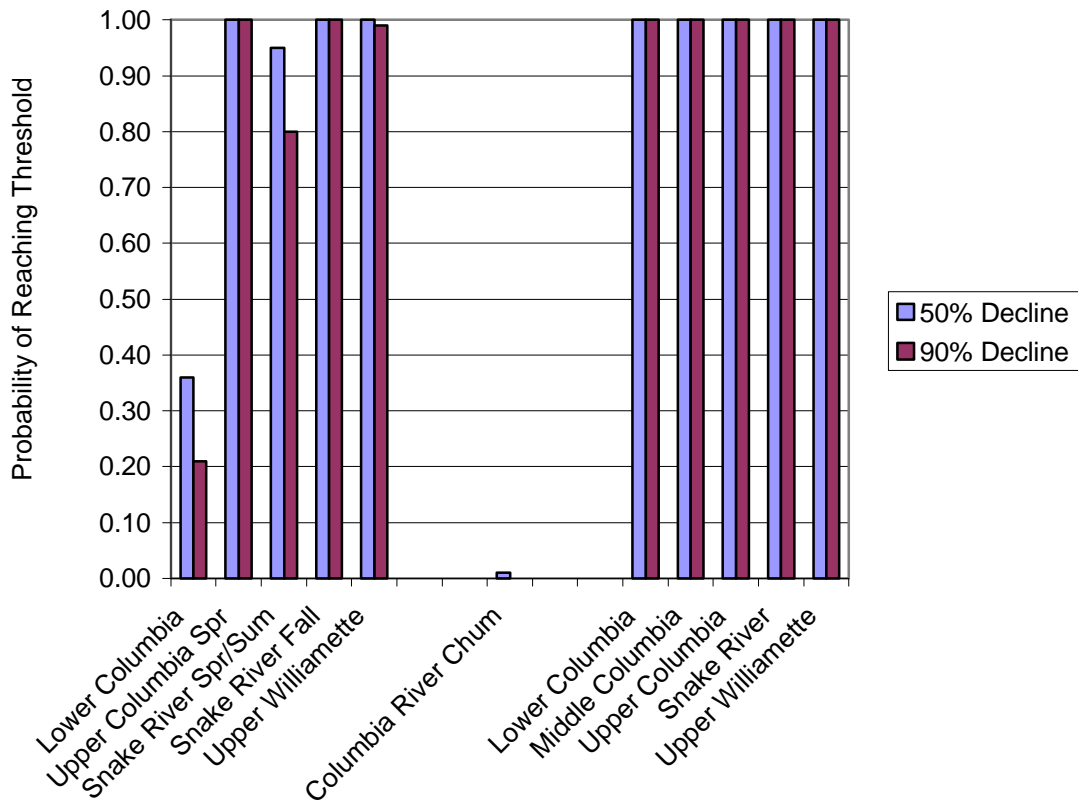


Figure IV-8. Probability that ESU is 50 or 90% below current levels at 100 years in the future. Note that this is not the probability that the ESU has dipped 50% or 90% below current levels within 100 years. This probability is better reflected by the most likely time to first hit a 50 or 90% decline. Note that ESUs with a positive lambda may have a short most likely time to first hit a 50 or 90% decline but a small probability that the stock is still 50 or 90% declined in 100 years.

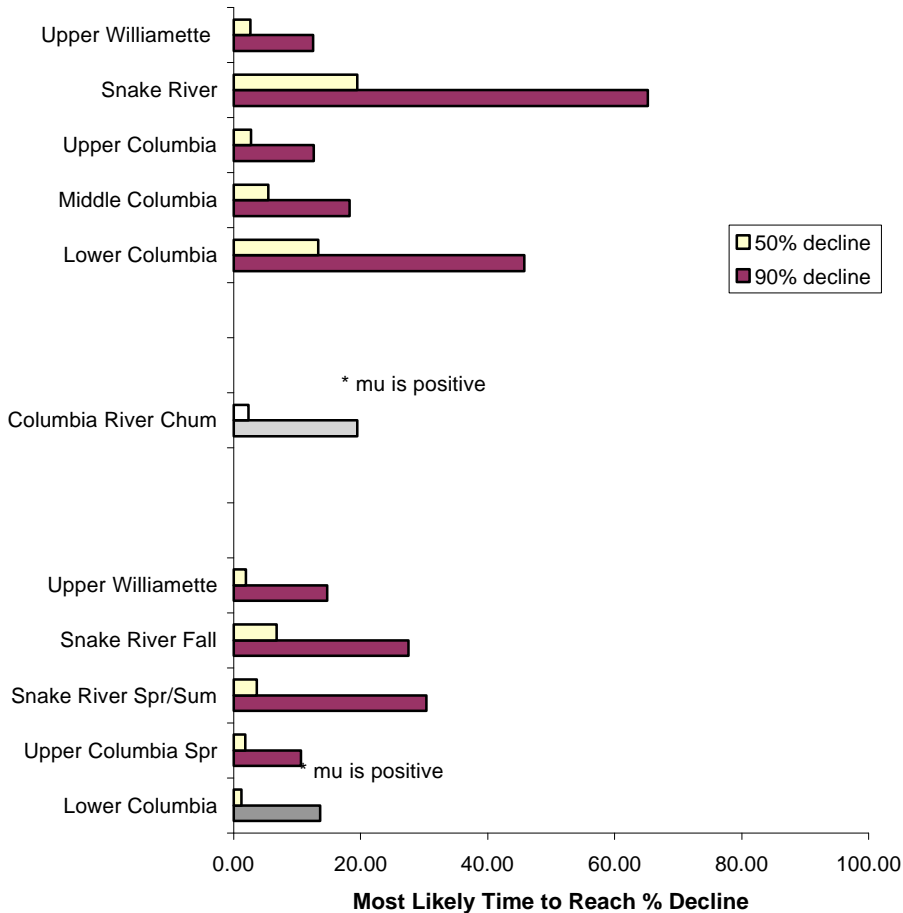


Figure IV-9. Most likely time to reach a 50% or 90% decline in the total population. This is the mode of the distribution of times to reach these declines and thus will tend to be lower than the mean time or median time. Note that the most likely time to reach these levels and the probability that the stock is 50% or 90% declined at year 24 or 100 are very different statistics. The first is the mean time to FIRST reach a 50% or 90% declined level while the second is the probability that at year 24 or 100 the stock is 50% or 90% declined. Stocks with positive lambda and high variance may have a high probability of hitting 50% or 90% declines early (due to high variance), but have a very low probability of being 50% or 90% declined in the long-term (because the overall population growth rate is positive).

Effect of hatchery fish on lambda

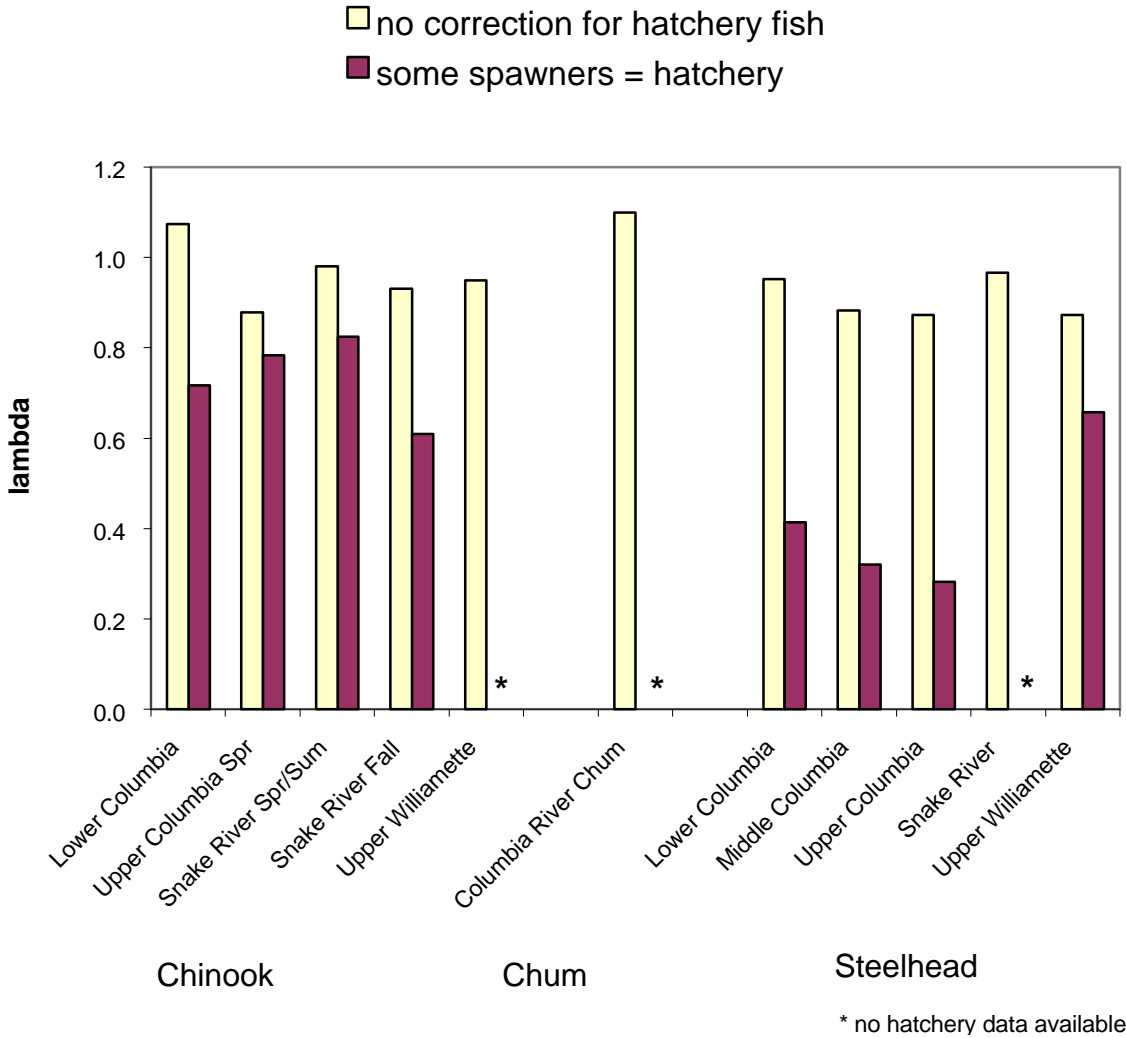


Figure IV-10. Estimates of the rate of population decline/growth for wild fish that accounts for the presence of hatchery fish in spawner counts. In one case (light bars), lambda is calculated without any correction for hatchery fish in spawner counts. In the other case (dark bars), hatchery fish are assumed to reproduce at the same rate as wild fish. Note that the potential influence of hatchery fish is greater for steelhead than for chinook ESUs.

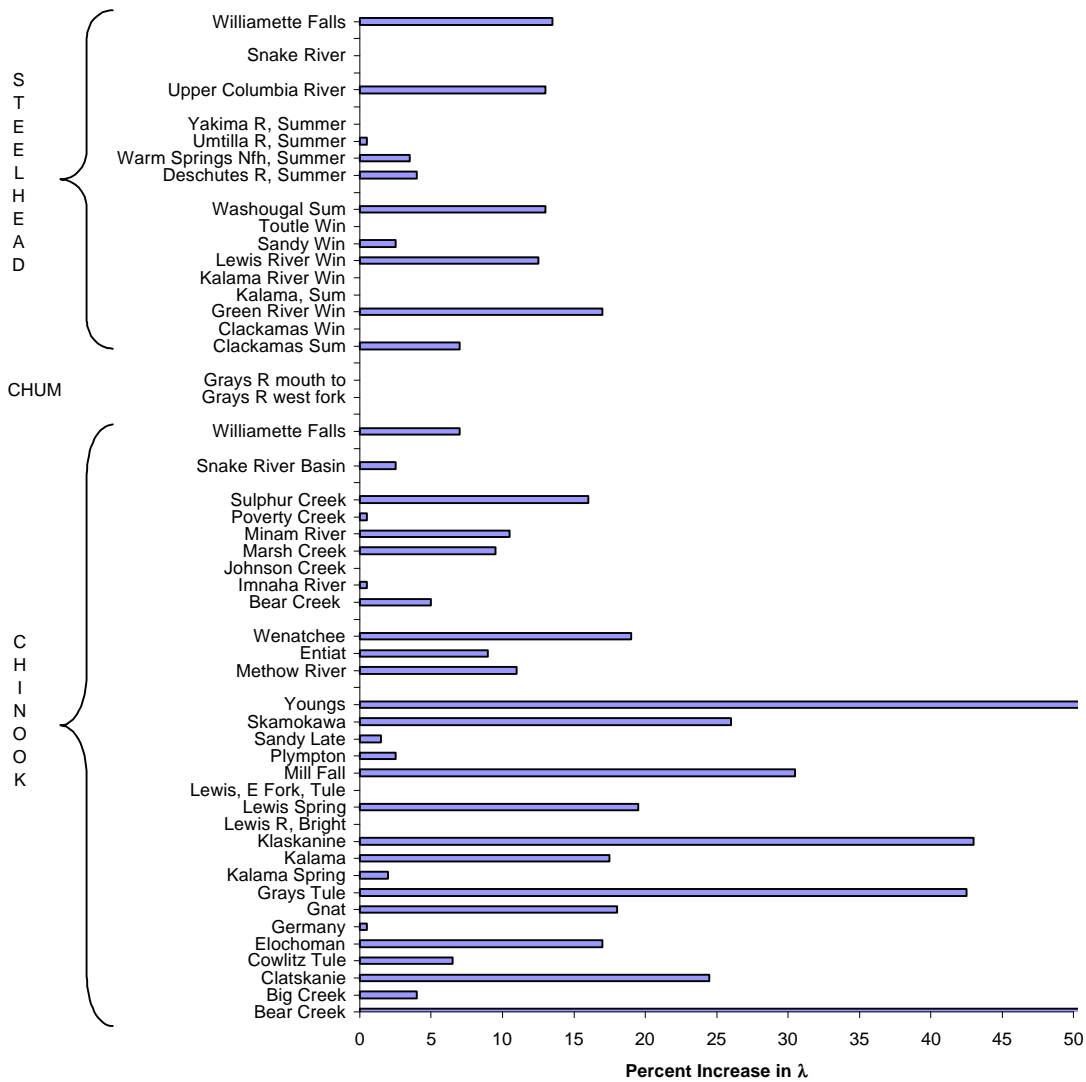


Figure IV-11. Percent increase in lambda required to reduce the risk of extinction (one fish in one generation) in 100 years to less than 5%.

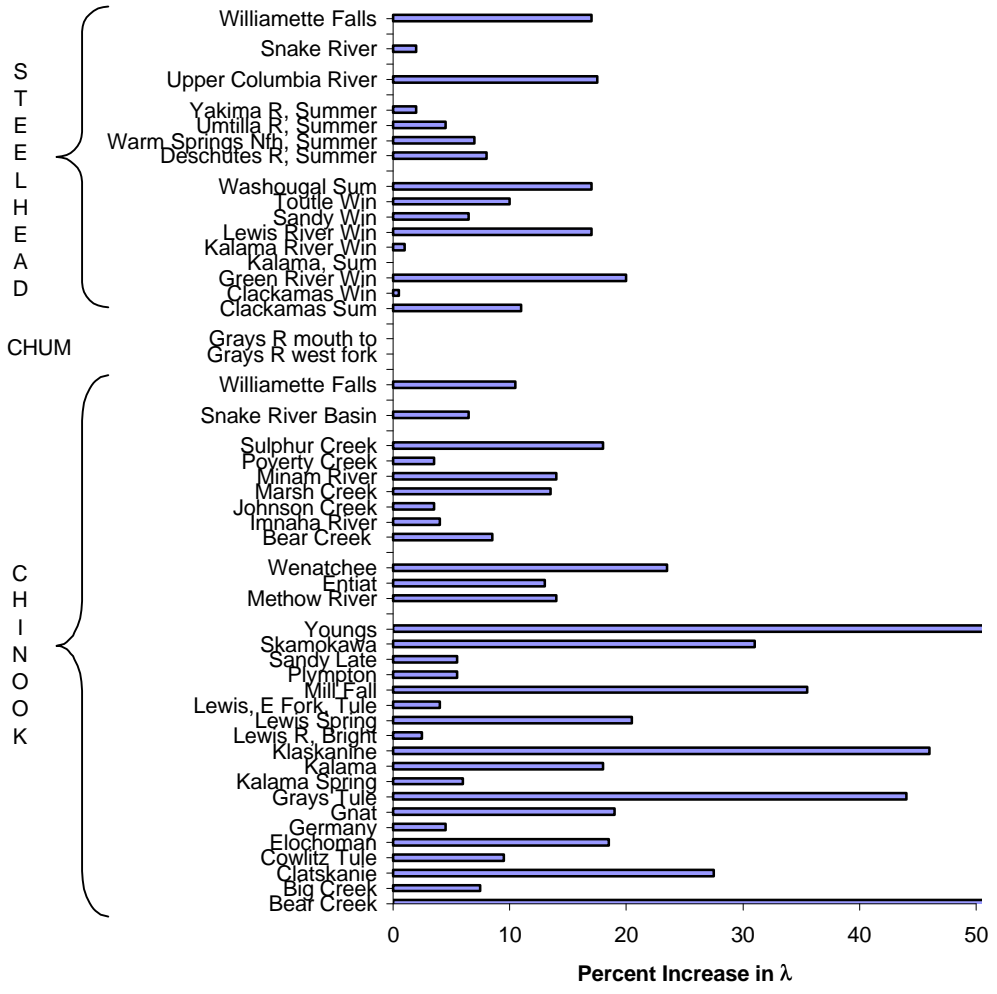


Figure IV-12. Percent increase in lambda required to reduce the risk to less than 5% that the stock declines to 90% of current levels in 100 years. This extinction risk measure is much less sensitive to errors in our estimate of the total number of spawners in the stock.

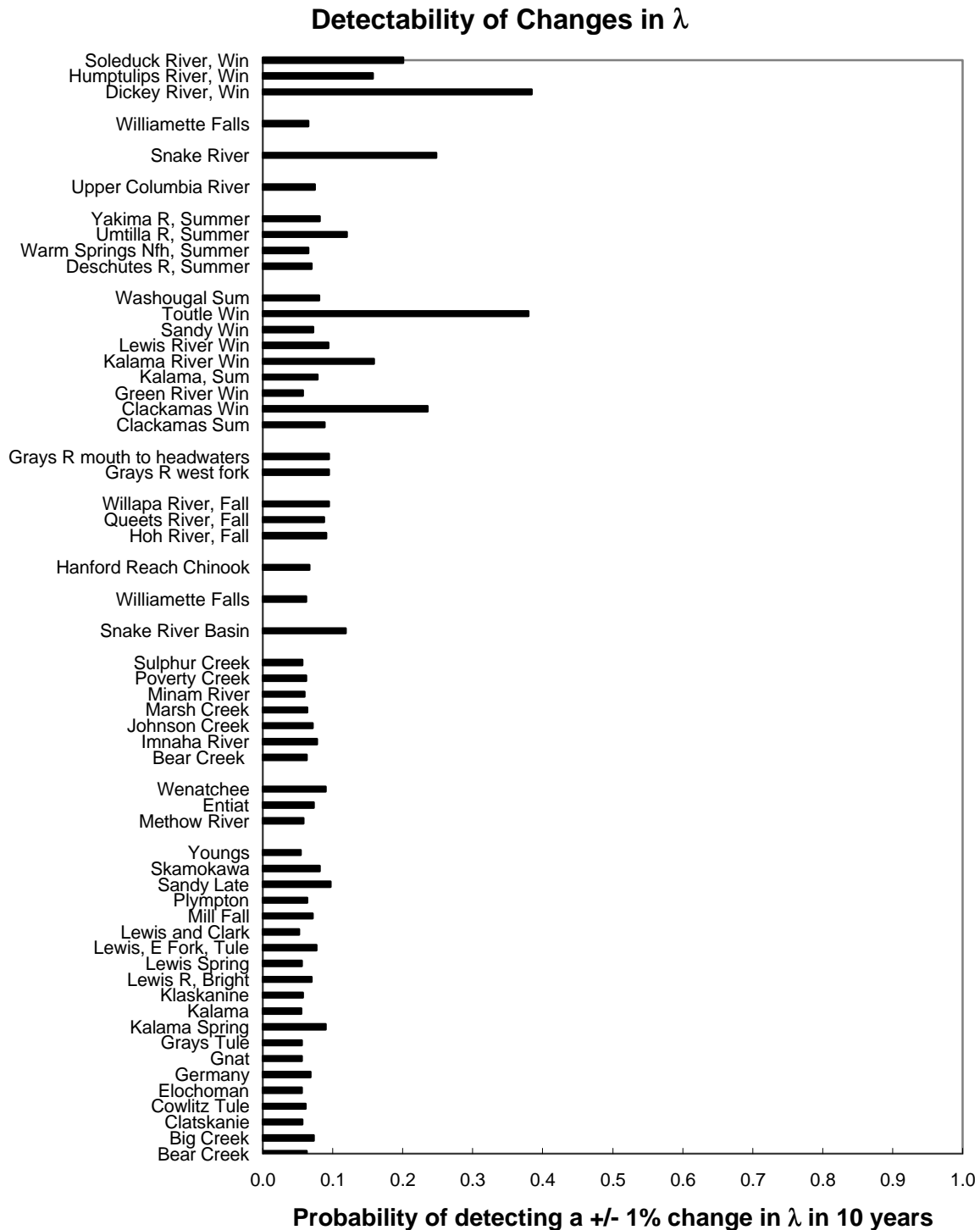


Figure IV-13. Detectability of management actions on Columbia River stocks. With an imposed change in population growth rate (I) of +/- 1%, what is the probability that in 10 years the projected population size differs from a population trajectory with I unchanged at the $p < 0.05$ level?

Detectability of Changes in λ

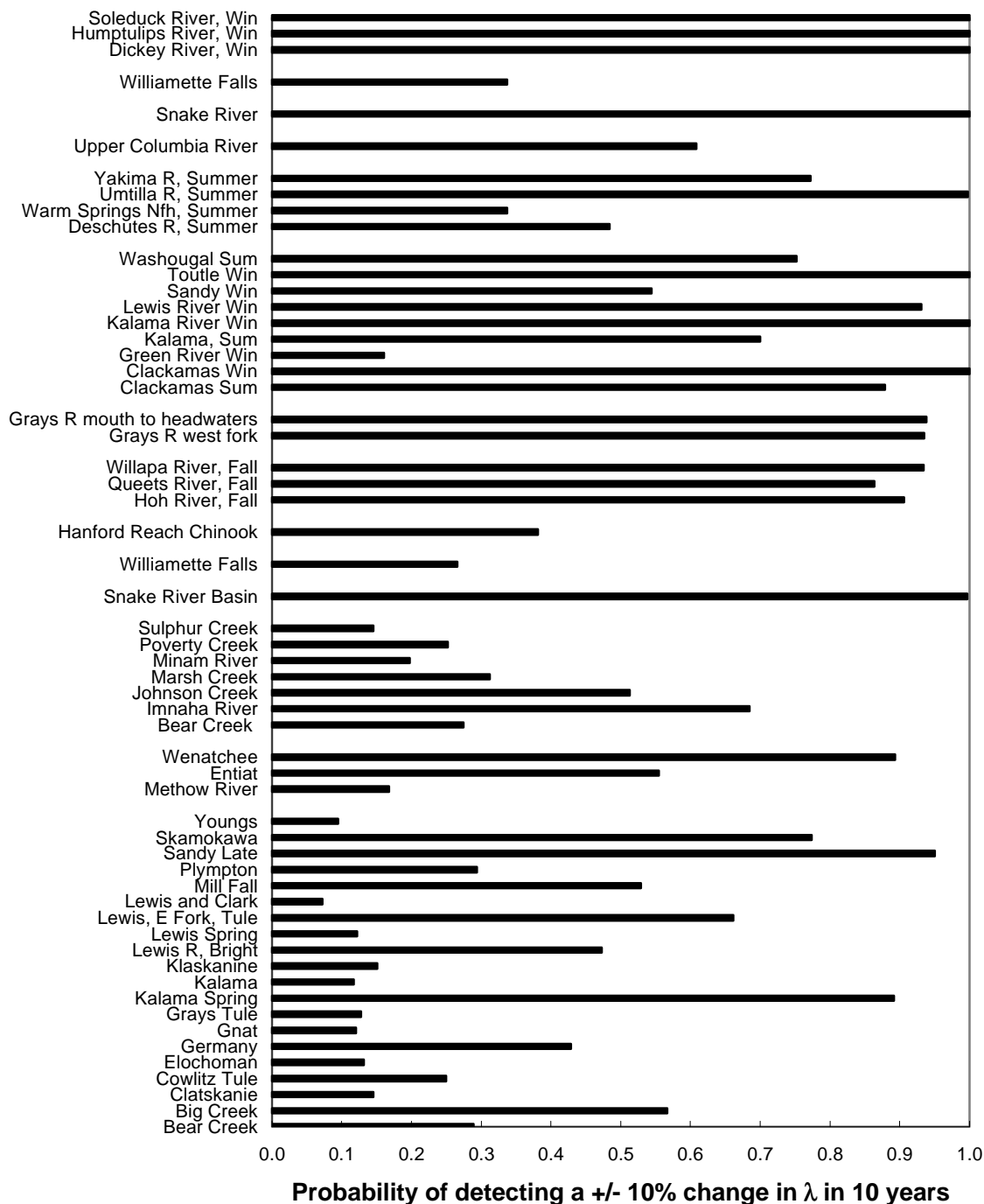


Figure IV-14. Detectability of management actions on Columbia River stocks. With an imposed change in population growth rate (I) of +/- 10%, what is the probability that in 10 years the projected population size differs from a population trajectory with I unchanged at the $p < 0.05$ level?

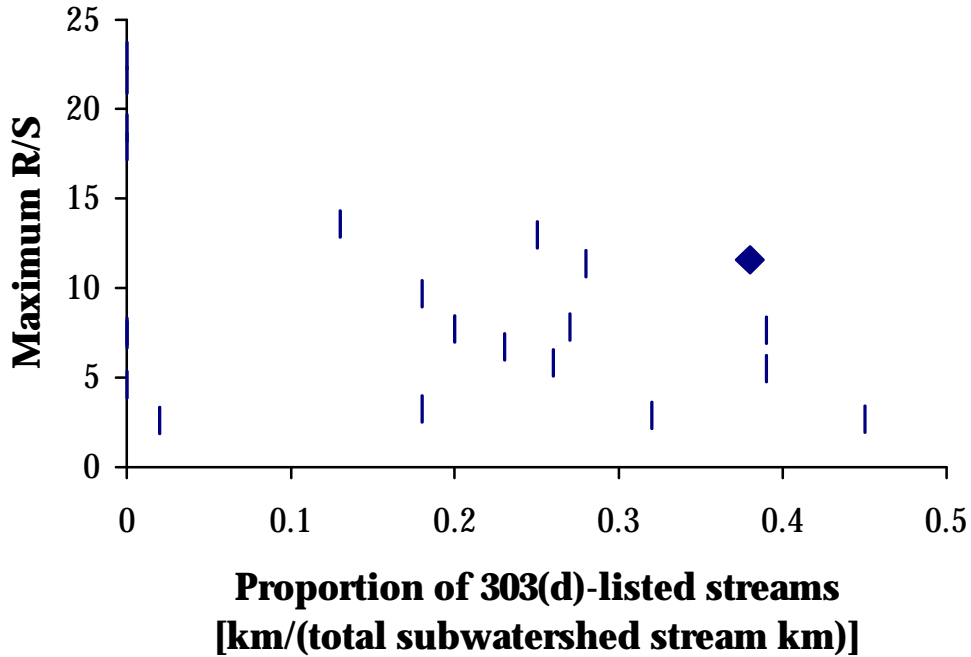
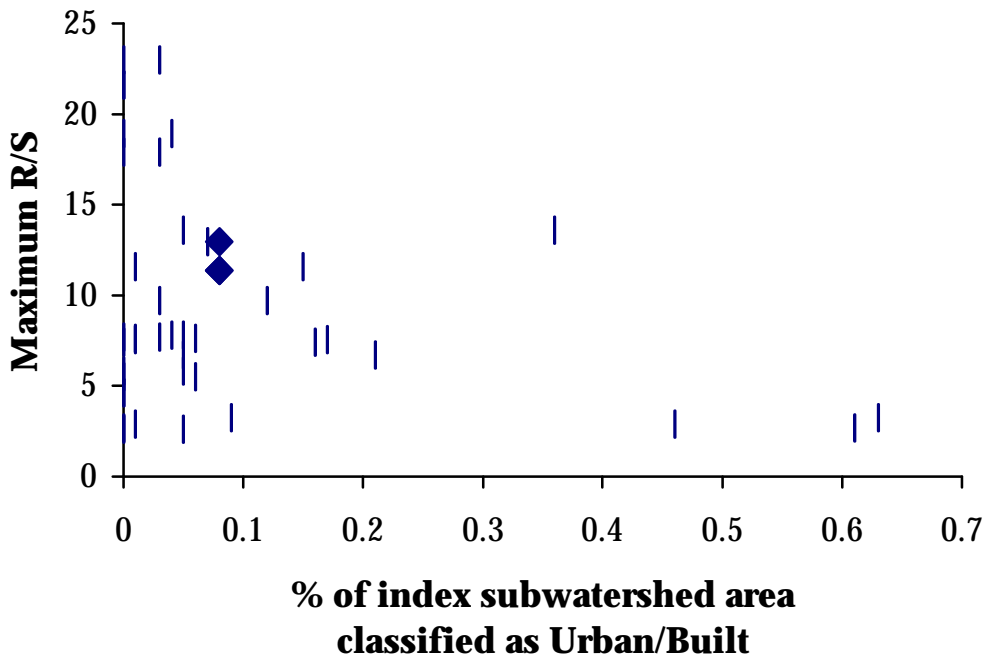


Figure IV-15. Plots of maximum *R/S* values, for all 22 index areas, against (a) percent of subwatershed area classified as urban/built, and (b) proportion of subwatershed streamlength listed under EPA 303(d). From Regetz (in review, Conservation Biology)

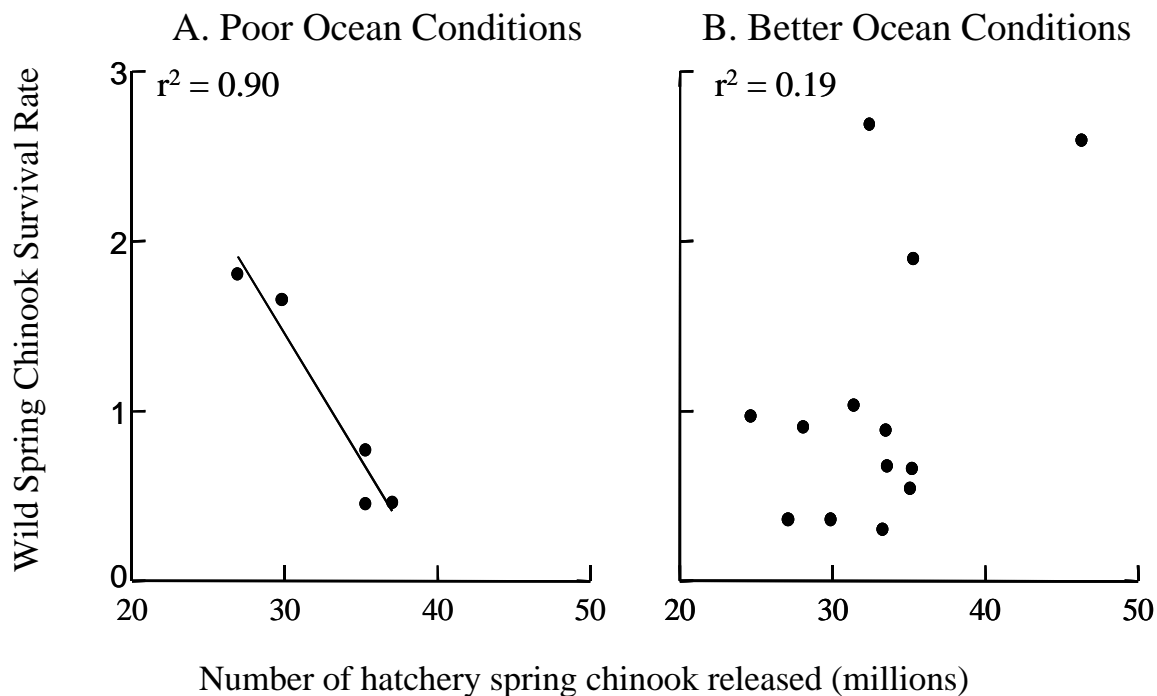


Figure IV-16. Survival rates of wild spring/summer Snake River Chinook salmon as a function of the total number of hatchery spring chinook released in the Columbia River Basin during (a) poor ocean conditions and (b) better ocean conditions. Data are from brood years 1975 to present.

Maximum Annual Population Growth Rate with Harvest Reductions/Moratoria

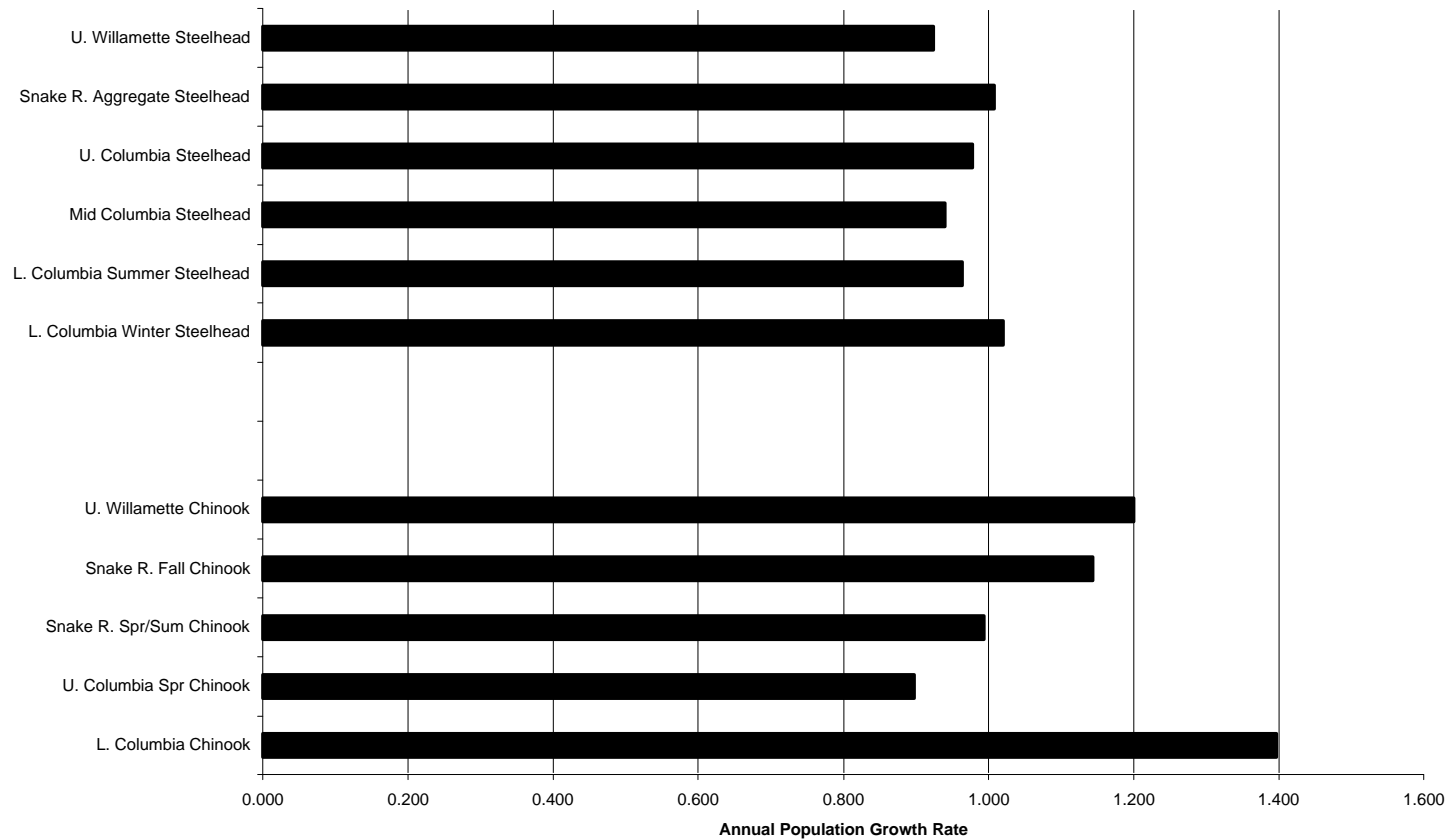


Figure IV-17. Predicted values of annual population growth rate (*I*), with no harvest, using aggregated ESU parameters. This is the maximum benefit that could be obtained through harvest restrictions or moratoria.

Maximum Percent Change in Pop. Growth Rate with Harvest Reductions/Moratoria

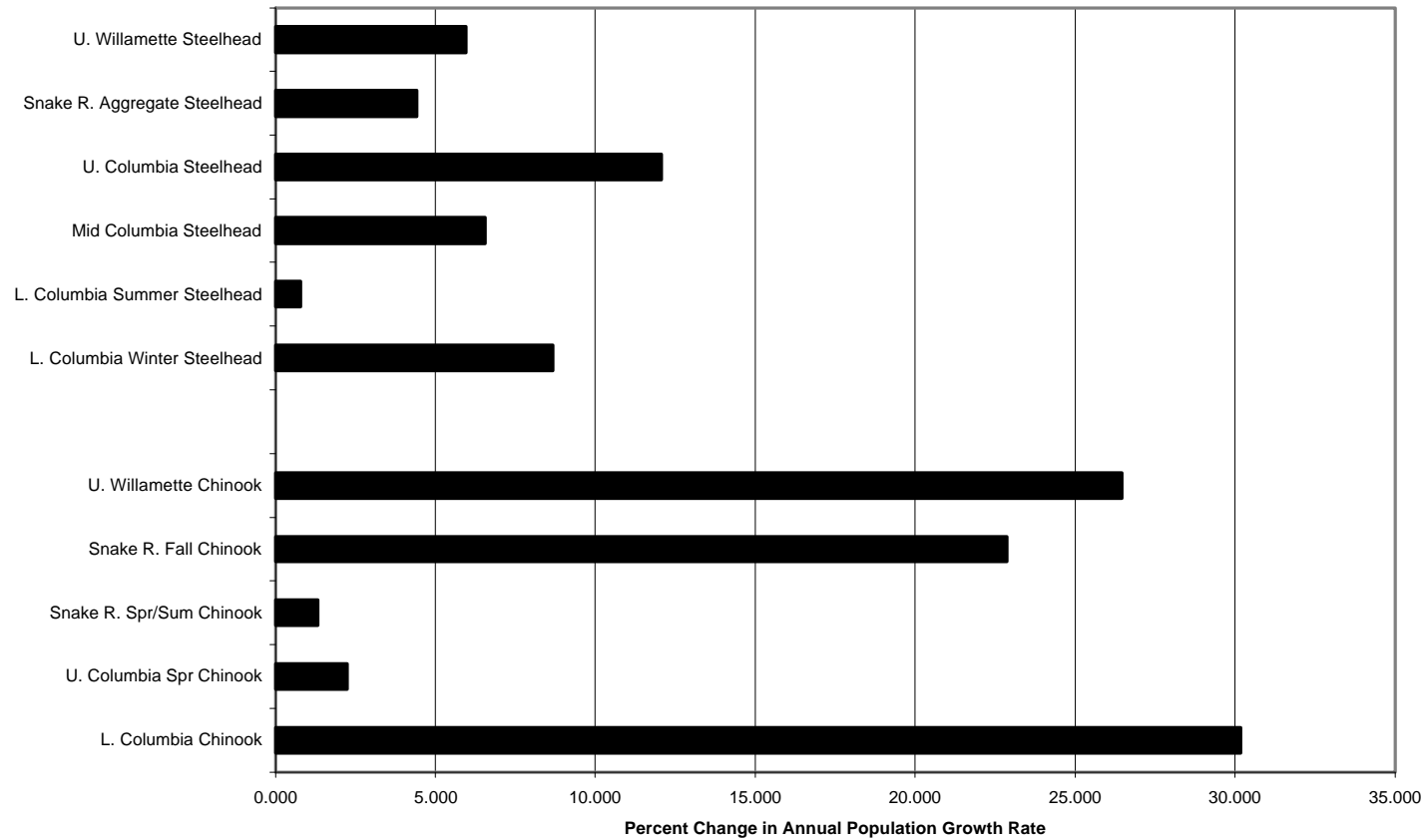


Figure IV-18. Predicted percent increase in annual population growth rate (*I*), with no harvest, using aggregated ESU parameters. This is the maximum benefit that could be obtained through harvest restrictions or moratoria.

V. DETAILED ANALYSES FOR SELECTED ESUs

V.A. Snake River Springs/Summer Chinook salmon

V.A.1. New data and the basic idea behind the analyses:

Most quantitative analyses regarding salmon in the Columbia River Basin, especially with respect to dam breaching, have dealt with Snake River spring/summer chinook salmon. In all cases, the analyses are conducted on seven “index stocks” (Table V-1), for which age-structured run-reconstructions have been completed. In this section we use these same index stocks, but include more recent data (through brood year 1994). The data for spring/summer chinook are based on redd counts expanded to estimate total numbers of spawners per reach for seven index stocks (Table V-1). Our results are reported for brood years 1980-1994 (or data spanning 1980-1999) or for brood years 1990-1994 (data spanning 1990-1999). We single out brood years 1990-1994 to address concerns that by treating the entire time period between 1980 and current, we might misconstrue a situation that has been progressively declining (that is recruits per spawner have been declining steadily during that time period).

Our goal in analyzing these matrices and related data for Snake River spring/summer chinook salmon is to gain insight into where the greatest opportunities for improving population performance exist. The measure of stock or population performance is annual rate of population change (I). Improvements are measured in the currency of changes in I . We do not view these exercises as models of future population growth, but rather as evaluations of likely improvements in population growth as a result of actions that take effect in particular stages of the lifecycle. These sorts of analyses are not possible for most Columbia River Basin ESUs because of an absence of detailed age-structured data. Even with these age-structured data, we do not feel it is advisable to simulate populations into the future. To do so would require estimates of carrying capacity and density-dependent feedbacks, for which we lack data. In addition, an explicit model of ocean cycles and their impacts on age-specific demography would be necessary – an enterprise that is well beyond our existing database.

For those who have read earlier CRI analyses of Snake River Spring/Summer Chinook salmon, it will be clear that these updated analyses are more complicated. Specifically, we present the standard deviation for numerical experiments, and perform many different permutations of numerical sensitivity experiments (using a wide variety of assumptions about baseline demography). Specifically, matrices were calculated and numerical experiments run for all seven index stocks in four different ways (for a total of 28 permutations). The four different approaches for estimating matrices are: (i) using run-reconstruction data for brood years 1980-1994, assuming survival in estuary is 7%, and then solving for first-year freshwater survival to balance Euler’s equation; (ii) using run-reconstruction data for brood year 1990-1994, assuming survival in estuary is 7%,

and then solving for first-year survival to balance Euler's equation; (iii) using run-reconstruction data for brood year 1980-1994 and solving for estuarine survival to produce the smolt-to-adult returns reported by PATH, and then solving for survival in the first year to balance Euler's equation; and (iv) using run-reconstruction data for brood year 1990-1994 and solving for estuarine survival to produce the smolt-to-adult returns reported by PATH, and then solving for survival in the first year to balance Euler's equation. Even though the numerical experiments were performed over a much wider range of baseline matrices than ever before, the results remain essentially unchanged. The conclusions at the end of this section are remarkably similar to the conclusions presented in earlier CRI documents, even though details and numbers have varied and new data have been added. This suggests that the conclusions are quite robust.

Table V-1. Number of Adult Spawners (S) (minus jacks) Estimated From Redd Counts and the Number of Adult Recruits (R) to the Spawning Grounds for Seven Stocks of Snake River Spring/Summer Chinook (Beamesderfer et al. 1998, and then recently updated by PATH, memo sent in January 2000). For these stocks, adults are age 4 & 5; jacks are age 3. Two entries (the number of Sulphur Creek spawners in 1984 and the number of Marsh Creek recruits in 1994) were changed from 0 to 1 fish.

yr	Marsh		Johnson		Imnaha		Bear Valley		Poverty Flats		Sulphur		Minam	
	S	R	S	R	S	R	S	R	S	R	S	R	S	R
80	16	178	55	130	183	497	42	260	163	318	12	44	43	220
81	115	190	102	151	453	598	151	236	187	323	43	300	50	538
82	71	228	93	125	590	476	83	413	192	218	17	150	104	170
83	60	472	152	387	435	550	171	1200	337	1074	49	598	103	489
84	100	56	36	107	557	141	137	89	220	209	0	57	101	156
85	196	86	178	81	699	99	295	140	341	246	62	116	625	183
86	171	102	129	205	479	104	224	229	233	717	385	252	357	167
87	268	54	175	106	448	52	456	149	554	423	67	38	569	58
88	395	273	332	433	606	339	1109	712	844	904	607	257	493	113
89	80	21	103	79	203	115	91	70	261	278	43	16	197	54
90	101	2	141	17	173	45	185	18	572	56	170	3	331	15
91	72	5	151	29	251	39	181	18	538	90	213	7	189	21
92	114	61	180	138	363	188	173	138	578	192	21	35	102	228
93	216	225	357	244	1178	146	709	499	866	429	263	163	267	198
94	9	0	50	31	115	54	33	43	209	97	0		22	65
95	0		20		97		16		81		4		45	
96	18		49		219		56		135		23		233	
97	110		236		474		225		363		43		140	
98	164		119		159		372		396		140		122	
99	0		49		282		72		153		0		96	

V.A.2. Testing for density-dependence in the index stock time series of recruits per spawner:

In section IV.A. we reported the results of tests for density-dependence across all twelve Columbia Basin ESUs, including Snake River spring/summer chinook salmon. At the aggregated ESU level, there was no evidence of density-dependence for spring/summer chinook salmon in the Snake River. We have also applied the same analyses to recruits per spawner data broken into the separate index stocks (again applying the Dennis and Taper 1994 methods); in this case, seven different sets of bootstrap simulations were performed, one for each index stock. For six of the seven index stocks, the null hypothesis of density-independent population dynamics cannot be rejected (Table V-2). The absence of density-dependence evident in Table V-2 runs counter to results reported in Schaller et al. (1999) possibly due to differences in techniques, time series length and the treatment of index stocks as a single or individual populations. Based on the Dennis and Taper (1994) analysis we conclude that it is reasonable to neglect density-dependence for the purpose of the numerical experiments we conduct regarding sensitivity analyses.

Table V-2. Parametric bootstrap likelihood ratio test for density-dependence for Snake River spring/summer chinook, where a is the y-intercept and b is the slope from a linear regression of ln(recruits per spawner) vs. the density of spawners. (* indicates statistically significant evidence of density-dependence.)

Stock	a1	var1	a2	b2	var2	Test statistic	Critical value
Marsh	-0.5640	2.6927	-0.3488	-0.0016	2.6669	0.1262	6.5125
Johnson	-0.1429	0.7736	0.1674	-0.0021	0.7393	0.6035	7.7120
Imnaha	-0.9043	0.8593	-0.2818	-0.0014	0.7289	2.3248	3.7596
Bear	-0.1395	1.5268	0.1475	-0.0011	1.4378	0.8046	8.3003
Poverty	-0.2380	0.9041	0.5637	-0.0020	0.7001	3.7886	7.2113
Sulphur	0.1717	4.5091	0.9741	-0.0058	3.5568	3.2129	7.4181
Minam	-0.2838	2.4894	1.1399	-0.0060	1.1756	14.5286*	7.0086

V.A.3 Estimating matrices for spring/summer chinook salmon

Published estimates for stage-specific survival and run-reconstructions were used to derive parameter estimates for Snake River Spring/Summer Chinook projection matrices. For the seven stocks of Snake River chinook examined here, adults return as 3-, 4-, or 5-year olds to spawn and die -- exceedingly few return at age 6 or older. In addition, these stocks are all stream-type salmon that spend two winters in freshwater before migrating to the ocean. The matrices only keep track of females -- we therefore implicitly assume a 1:1 sex ratio of eggs and equal survival probabilities of males and females. The basic structure of demographic matrices for spring/summer chinook salmon in the Snake River is as follows:

	1	2	3	4	5
1			$(1-m)s_1b_3m_3/2$	$(1-m)s_1b_4m_4/2$	$(1-m)s_1m_5/2$
2	s_2				
3		s_3			
4			$(1-b_3)s_4$		
5				$(1-b_4)s_5$	

where s_x is the probability of survival of females from age $(x-1)$ to age x , m_x is the number of eggs per female spawner of age x , b_x is the propensity of females of age x to breed, and m is mortality of adult females during their migration upstream. Estimates of m_x , assuming constant fecundity for all three adult age classes, are from Myers *et al.* (1998; Table V-3). For mortality during upstream migration,

$$m = 1 - (ps(1 - harvest_{sb})BontoBasin(1 - harvest_{ms})) \quad [5.1]$$

where ps , the prespawn survival, is set to 0.9 in all years, $BontoBasin$ is survival of unharvested spawners from Bonneville dam to their spawning basin, $harvest_{sb}$, the rate of harvest in the subbasin, was 0 for recent years, and $harvest_{ms}$ is harvest in the mainstem of the Columbia River (Beamesderfer *et al.* 1998; Table V-3). Annual age frequencies of spawners (Beamesderfer 1998) were used to calculate b_x . Because these data are for both males and females, the age frequencies were corrected, using sex ratio at age data from Hall-Griswold and Cochnauer (1988), White and Cochnue (1989), and Elms-Cockrom (1998) to find the proportion of the total returning adults of age x that are female (f_x) (Table V-3).

Table V-3. Mean fecundity, upstream survival, and percent female at age for the seven index stocks of Snake River spring/summer chinook using data from 1980-1999.

Stock	mean fecundity	mean Bonto Basin	mean harvest _{ms}	% female ₃	% female ₄	% female ₅
Marsh, Bear, Sulphur	5607	0.581	0.079	0	49	67
Johnson & Poverty	4100	0.681	0.029	11	28	58
Imnaha*	4927	0.631	0.054	11	28	58
Minam*	4086	0.581	0.079	11	28	58

*No data regarding age distribution by sex were available for Imnaha and Minam. We used sex ratios from Johnson & Poverty because Imnaha & Minam have hatchery fish, which are known to spawn at younger age than wild fish.

Table V-4. Estimates for various time periods and for different future scenarios. Estimates of current survival during downstream migration and the proportion of smolts transported in barges were obtained from Marmorek (1998).

	1980-1999	1990-1999	1977-1979	Improved hydro	Draw-down
s_d	0.5817	0.7285	0.0946	0.7664	0.6066
pt	0.1448	0.2016	0	0.2400	0

Survival during transport (s_b) was assumed to be 0.98. Few data are available with which to estimate estuarine and early ocean survival, but survival during the first year in the ocean is thought to generally be between 2-10% (Nickelson 1986, Pearcy 1992); for our baseline matrices, we either set s_e to 7%, or solved for estuarine survival to produce the SARs reported by PATH. In other words we either calculated s_2 directly (assuming the 7% estuarine survival value) or solved for its value in order to produce PATH SARs. In either case, s_2 can be decomposed as follows:

$$s_2 = ((1-pt) * s_d + pt*s_b)* s_e. \quad [5.2]$$

There are no direct estimates of adult survival in the ocean. As in Ratner et al. (1997), we set $s_3 = s_4 = s_5 = 0.8$.

To find the f_x 's for females only, the age frequencies for each year were multiplied by the % female_x, re-scaled so the frequencies summed to one, and averaged across the time series (Table V-5). Snake River spring/summer chinook salmon are not known to breed beyond age 5, so b_5 was set to 1. b_3 and b_4 were estimated by solving a set of simultaneous equations:

$$f_x = \frac{b_x l_x}{\sum b_i l_i} \text{ for } x = [1,5], l_x = \prod p_x, p_1 = s_1, \quad [5.3]$$

$$\text{and } p_x = (1 - b_{x-1})s_x \text{ for } x > 1$$

(Ratner et al. 1997; Table V-5). Assuming that productivity does not depend on spawner density, the average $\ln(R/S)$ from 1980-1994 was used as the estimate of productivity (Table V-5). Survival from egg to LGR (s_1) was found by simultaneously solving Euler's equation (Ratner et al. 1997; Table V-5),

$$(1 - m) \sum l_x \left(\frac{m_x}{2}\right) b_x I^{-x} = 1, \quad \text{with } I^T = e^{avg.\ln(R/S)} \quad [5.4]$$

$$\text{and } T = (1 - m) \sum x l_x \left(\frac{m_x}{2}\right) b_x I^{-x}$$

Table V-5. Age distributions of females, age-specific propensity to breed, average productivity, and survival from egg to one year old for seven index stocks of Snake River spring/summer chinook.

Stock	avg. f_3	avg. f_4	avg. f_5	b_3	b_4	average $\ln(R/S)$	s_1
Marsh	0	0.256	0.744	0	0.216	-0.564	0.018
Bear	0	0.249	0.751	0	0.210	-0.140	0.027
Sulphur	0	0.247	0.753	0	0.208	0.172	0.037
Johnson	0.015	0.228	0.757	0.010	0.194	-0.143	0.030
Poverty	0.019	0.217	0.764	0.013	0.185	-0.238	0.027
Imnaha	0.016	0.360	0.624	0.011	0.316	-0.904	0.013
Minam	0.010	0.474	0.516	0.007	0.423	-0.284	0.031

The resulting seven baseline matrices representing average current conditions using the 7% estuarine survival approach are given in Table V-6. An additional set of matrices based only on data from 1990 onward is given in Table V-7. (The matrices found by solving for estuarine survival to produce PATH SARs are provided on the CRI website.) The sensitivity and numerical experiments are robust to all 28 permutations of the matrices examined; thus, difference in the matrix details among the 28 permutations do not greatly affect the conclusions. As explained above, growth rates associated with the matrices in Tables V-6 and V-7 are determined by the average recruits per spawner rates. Although median recruits per spawner, or the geometric mean matrix, could be used, the results discussed below are not qualitatively altered by these alternative methods.

For a detailed population viability analysis, estimates of temporal variation for each matrix entry, as well as some estimate of how the different matrix entries co-vary, would be warranted. There is little chance that such detailed data will be forthcoming for any salmonid stock over the next 10 years. Arguably, it is also unlikely that much would be gained from these more detailed data. The most striking results from these matrix syntheses of run-reconstruction data are:

- 1.) Regardless of the time period (1980-1999 or 1990-1999), several index stocks appear to be declining at rates in excess of 10% per year. Since the matrices and population growth rates do NOT incorporate the effects of environmental variation, which would increase the rate of decline considerably, the Snake River spring/summer chinook salmon stocks are clearly in substantial peril.
- 2.) The rate of decline has increased in recent years. For example, the average rate of decline for brood years 1990-1994 is an annual decrease of 24%, whereas the average rate of decline for brood years 1980-1994 is a more modest 6% annual decline.
- 3.) The relative performance of different index stocks varies with the time period being analyzed and is not consistent through time. A striking example of this is Sulphur Creek. This stock stands out as the only one showing an annual growth rate greater than 1 for the long time-frame of brood year 1980-1994; yet, over the time period corresponding to brood year 1990-1994, it suffers an annual decline of 32% which is lower than all but one of the index stocks.

Table V-6. Parameterized matrices and population growth rates (λ) for seven index stocks of Snake River spring/summer Chinook salmon based on data from 1980-1999.

Marsh:	1	2	3	4	5	$\lambda = 0.888$
1	0	0	0	5.162	23.914	
2	0.044	0	0	0	0	
3	0	0.800	0	0	0	
4	0	0	0.800	0	0	
5	0	0	0	0.627	0	

Johnson:	1	2	3	4	5	$\lambda = 0.970$
1	0	0	0.370	7.068	36.401	
2	0.044	0	0	0	0	
3	0	0.800	0	0	0	
4	0	0	0.792	0	0	
5	0	0	0	0.645	0	

Imnaha:	1	2	3	4	5	$\lambda = 0.824$
1	0	0	0.189	5.243	16.596	
2	0.044	0	0	0	0	
3	0	0.800	0	0	0	
4	0	0	0.791	0	0	
5	0	0	0	0.547	0	

Bear/Elk:	1	2	3	4	5	$\lambda = 0.971$
1	0	0	0	7.666	36.568	
2	0.044	0	0	0	0	
3	0	0.800	0	0	0	
4	0	0	0.800	0	0	
5	0	0	0	0.632	0	

Poverty:	1	2	3	4	5	$\lambda = 0.951$
1	0	0	0.424	6.134	33.130	
2	0.044	0	0	0	0	
3	0	0.800	0	0	0	
4	0	0	0.790	0	0	
5	0	0	0	0.652	0	

Sulphur:	1	2	3	4	5	$\lambda = 1.037$
1	0	0	0	10.381	49.942	
2	0.044	0	0	0	0	
3	0	0.800	0	0	0	
4	0	0	0.800	0	0	
5	0	0	0	0.634	0	

Minam:	1	2	3	4	5	$\lambda = 0.939$
1	0	0	0.204	12.725	30.055	
2	0.044	0	0	0	0	
3	0	0.800	0	0	0	
4	0	0	0.795	0	0	
5	0	0	0	0.461	0	

Table V-7. Parameterized matrices and population growth rates (λ) for seven index stocks of Snake River spring/summer Chinook salmon based on data from 1990-1999.

Marsh:	1	2	3	4	5	$\lambda = 0.675$
1	0	0	0	1.356	5.313	
2	0.054	0	0	0	0	
3	0	0.800	0	0	0	
4	0	0	0.800	0	0	
5	0	0	0	0.596	0	

Johnson:	1	2	3	4	5	$\lambda = 0.815$
1	0	0	0.131	2.637	12.984	
2	0.054	0	0	0	0	
3	0	0.800	0	0	0	
4	0	0	0.792	0	0	
5	0	0	0	0.638	0	

Imnaha:	1	2	3	4	5	$\lambda = 0.748$
1	0	0	0.124	3.369	8.709	
2	0.054	0	0	0	0	
3	0	0.800	0	0	0	
4	0	0	0.789	0	0	
5	0	0	0	0.491	0	

Bear/Elk:	1	2	3	4	5	$\lambda = 0.812$
1	0	0	0	2.913	12.806	
2	0.054	0	0	0	0	
3	0	0.800	0	0	0	
4	0	0	0.800	0	0	
5	0	0	0	0.618	0	

Poverty:	1	2	3	4	5	$\lambda = 0.757$
1	0	0	0.096	1.719	9.140	
2	0.054	0	0	0	0	
3	0	0.800	0	0	0	
4	0	0	0.792	0	0	
5	0	0	0	0.650	0	

Sulphur:	1	2	3	4	5	$\lambda = 0.681$
1	0	0	0	1.292	5.509	
2	0.054	0	0	0	0	
3	0	0.800	0	0	0	
4	0	0	0.800	0	0	
5	0	0	0	0.612	0	

Minam:	1	2	3	4	5	$\lambda = 0.849$
1	0	0	0.101	6.804	15.569	
2	0.054	0	0	0	0	
3	0	0.800	0	0	0	
4	0	0	0.795	0	0	
5	0	0	0	0.450	0	

V.A.4. Where in the Spring/Summer Chinook Life Cycle are the Greatest Opportunities for Recovery?

The relative value of potential changes in various demographic rates was assessed in two ways. First, the elasticity of the population growth rate to small changes in each demographic parameter was calculated. By this measure, the most important parameter is the survival of adults in the ocean (Figure V-1). This is not a particularly surprising finding, since survival of individuals near the age of reproduction is generally among the most important life history parameters for organisms (Caswell 1989). This occurs because elasticity depends, in part, on reproductive value, which is a measure of an individual's contribution of offspring to future generations. In general, individuals near the age of reproduction have high reproductive value because they have made it through the long period of lowest survival and are just about to contribute offspring.

The second way in which we compared the importance of various demographic rates was by calculating the percent change in the population growth rate (I) that would be achieved if we could save 1 out of every 10 fish that currently die at each lifestage. By this measure, the most important parameters are survival through the first year of life (s_1) and survival in the estuary and early ocean (s_e) (Figure V-2). For the 1980-1999 matrices, a 10 percent reduction in mortality during the first year of life is predicted to result in a 41.5 percent increase in I , on average across the seven stocks. A 10 percent reduction in mortality occurring as smolts enter the estuary and during their first winter in the ocean (s_e) is expected to result in a 19.8 percent average increase in I . Changes in other parameters have a much smaller effect on I than changes in s_1 or s_e (Figure V-2). This result is driven in part by the current low values of first year and estuarine survival; therefore, saving 1 out of every 10 fish that currently die at these stages would mean saving relatively more individual fish. The results displayed in figure V-2 are the average across all seven index stocks, but there is very little variation in this basic pattern among stocks (note the standard deviations are also shown in the figure). These same analyses were repeated for the matrices estimated using the PATH SARs as a starting point, and the pattern is not qualitatively changed (Figure V-3).

It is important to note that both of these sensitivity measures predict that the survival of in-river migrants (s_d) and the proportion of fish transported in barges (pt) exert little influence on the rate of population growth. In other words, if our estimates of current demographic rates are correct, we would expect little payoff (in terms of improved population growth) for further improvements in fish transportation or fish guidance systems. This is not to say that fish transportation or improvements in fish guidance systems to date have not been important. In fact, it is quite likely that Snake River Spring/Summer Chinook salmon would currently be extinct if efforts to transport smolts and other improvements to the hydropower system had not been made (see next section). In addition, it is also possible that further improvements in hydrosystem operation that influence survival below Bonneville Dam (because they enhance individual fitness or alter run-timing in beneficial ways) could have substantial payoffs.

V.A.5. Numerical Experiments as a Means of Evaluating Management Options With Respect to Spring/Summer Chinook Salmon

The first impression created by the above sensitivity analyses is that earlier reductions in harvest and historical improvements to the migration corridor have been unimportant, but this is not the case. Rather, these analyses indicate that further management actions aimed at harvest and downstream survival for spring/summer chinook salmon (excluding dam breaching) are not likely to be that helpful, even though past actions have been crucial. However, so much has already been done with these two management levers that stocks are now in a region of diminishing returns. To show this clearly, we conducted the following numerical experiments. First, harvest and all other demographic rates were held at their current values, but transportation was assumed to be non-existent ($pt = 0$) and other migration corridor rates impacted by dams were assumed to have been unaffected by improvements, including altered flow regimes and the addition of turbines. Specifically, s_d and *BontoBasin* (survival of adult migrants from the Bonneville Dam back to the basin of origin) were estimated from 1977-1979 run reconstruction data. The period from 1977-1979 was chosen because all of the currently existing dams were in place by then, but more recent improvements in dam operations had not yet occurred. During the late 1970s (and averaged across the seven stocks), s_d was only 0.0946 (less than the 1990-99 average of 0.2016) and the *BontoBasin* conversion rate averaged 0.56 (less than the 1990-1999 average of 0.64). This numerical simulation demonstrates that if survival through the hydropower system had remained at the low levels of the late 1970s and estuarine survival backcalculated from PATH-estimated SARs is reasonable, in the absence of transportation of smolts, populations of spring/summer chinook salmon in the Snake River would likely have already gone extinct (since the estimated annual decline assuming unimproved hydrosystem passage conditions is over 50 percent population loss each year; see Figure V-4). One obvious question is whether transportation or bypass systems could ever be improved to such an extent that, by themselves, these improvements would adequately reduce extinction risks. The answer is no. In particular, even if every smolt made it successfully to the estuary, annual population growth would increase on average by under 6 percent (Figure V-4). To see the extent to which a 6% increase in lambda “comes up short”, we can examine Figure IV-11, which presents the %increase in lambda required to reduce extinction risk in 100 years to less than 5%. Noting that Sulphur Creek requires a 16% increase in λ to reduce the extinction risk below this 5% value, Minam River requires a 10.5% increase in lambda and Marsh Creek requires a 9.5% increase in lambda, it is clear that we might expect to lose several stocks even if we could elevate downstream in-river survival to 100%. In short, perfect downstream survival, by itself, would not be enough to prevent extinction, given the mortality suffered during other life stages of spring/summer chinook salmon.

In a similar vein, another numerical experiment was performed with all rates at current levels except harvest; harvest rates typical of 1960-1970 were imposed (average mainstem harvest rate = 0.394, average subbasin harvest = 0.142 compared to 1990-1999 average mainstem harvest = 0.053, subbasin harvest = 0). Under harvest rates from the 1960s (and keeping all of the hydrosystem passage improvements and

transportation in place), it appears that population growth would be suppressed to a level that would also yield certain extinction (because I is substantially less than 1; see second bar from the left in Figure V-4). In sum, without harvest reductions and hydropower system improvements made over the last 20 years, spring/summer chinook salmon would probably have gone extinct by now (Figure V-4). However, given current rates, neither elimination of harvest or further improvements in direct downstream survival, by themselves, are likely to sufficiently improve population growth of spring/summer chinook (see the two rightmost bars in Figure V-4).

Another key question is **whether dam breaching, by itself, is likely to recover spring/summer chinook salmon populations?** To answer this question, it was assumed that breaching would have three main effects: 1) altered downstream survival (using the rates assumed by Marmorek (1998) for the breaching option), 2) possibly improved upstream survival (here we simulated four levels of potential improvement in upstream survival rates), and 3) possibly improved survival below Bonneville Dam because differential delayed transportation mortality and/or extra mortality would no longer be an issue. Differential delayed transportation mortality is related to reduced survival of transported fish compared to in-river migrants below Bonneville Dam, whereas extra mortality is a hypothesized reduction in the survival of both transported fish and in-river migrants that may be attributable to dams. Debate about the importance of these post-Bonneville effects of dams has been highly contentious, and data with which to estimate these parameters are generally poor. We, therefore, examined a broad range of potential improvements in survival below Bonneville Dam that could potentially occur after dam drawdown.

The results are summarized in Figure V-5. This figure is complicated, but by working through one of its panels, its interpretation becomes clear. Suppose that after drawdown, upstream survival is increased by 30%. That increase would produce an upstream survival of greater than 93%, which is probably as high as could be achieved. Then given that upstream survival, we can calculate how much λ would be increased assuming different ranges of improvement in early ocean/estuarine survival. To reduce extinction risk of Sulphur Creek to less than 5% over the next 100 years requires a 16% increase in λ or a doubling (100% increase) in estuarine/early ocean survival. This is precisely why it seems unduly optimistic to think that dam breaching by itself could adequately mitigate the extinction risk faced by spring/summer chinook salmon. For reference, a 20 percent improvement in s_e corresponds to a D approximately equal to 0.8, a 60 percent improvement in s_e corresponds to $D = 0.5$, and a 160 percent improvement corresponds to $D = 0.2$.

When the discussion is phrased in this manner, the key question is obvious: how much would dam breaching increase survival of downstream migrants, older fish in the estuary, and upstream migrants compared to current conditions? All of the possibilities can become overwhelming in their many combinations. One way of achieving some clarity is to consider one drawdown scenario that we feel is optimistic. For this scenario, the improvements associated with breaching were assumed to result in a 15 percent improvement in Bonneville to Basin survival, a downstream survival of 62 percent, and

an increase in estuarine survival of 60 percent (which roughly corresponds to a $D = 0.5$). To put this drawdown scenario in context, it was compared with an alternative scenario combining zero-harvest plus a 25 percent improvement in first year survival (s_1) due to habitat improvements. Viewed in this light, drawdown and the habitat/harvest actions are roughly equivalent in their effect on population growth, and neither, by themselves, is likely to recover Snake River chinook salmon (Figure V-6). One weakness of this analysis is that dam breaching may alter additional components of the life cycle, beyond the three parameters explored here. One possibility is that breaching would result in healthier fish that would experience reduced mortality not only in the estuary but in all subsequent years of adult life in the ocean. Alternatively, dam drawdown might result in increased habitat availability and possible improvements in s_1 (because of lower predation rates when reservoirs are drained). Dam breaching might also alter patterns of nutrient cycling and replenishment that, in turn, influence productivity.

V.A.6. Demography and dam breaching.

The major uncertainty for the CRI analyses is the biological feasibility of using particular management actions to achieve sufficient demographic improvements. The possibilities of recovering Snake River spring/summer chinook salmon through dam breaching, improved fish passage, habitat improvements and other means have been the topic of much recent discussion. Yet, data to inform this discussion are lacking. We do know that direct downstream survival cannot be increased enough to achieve significant gains in annual population growth rates. We also know that harvest rates are so minor that further harvest reductions are unlikely to substantially enhance rates of population growth. In contrast, the demographic consequences of virtually every other management action are uncertain.

Although dam breaching is a major option likely to offer substantial gains, it is unclear how substantial these gains would be. Some have argued that the magnitude of improvements due to dam breaching can be assessed by comparing upstream and downstream stocks (Schaller et al. 1999). Although this approach is sensible, it has several shortcomings that limit its effectiveness. Chief among the difficult issues facing such assessments is determining what constitutes a “control” for the “treatment” of the hydropower system. Schaller and colleagues use fish stocks residing in lower Columbia River basin, an area with fewer dams, as controls for stocks from upstream regions of the basin with greater numbers of dams. Unfortunately, because they are clumped rather than interspersed, stocks within regions are pseudoreplicates, making the separation of location from treatment effects problematic (Hurlbert 1984, Zabel and Williams in press, Levin in review). Additionally, stocks from different regions are genetically distinct and occupy watersheds of differing potential productivity (Myers et al. 1998, Regetz in review). The presence of differences between down and upriver stocks, therefore, is inconclusive because differences among regions may arise from many factors that vary among locations, only one of which is the number of dams.

Another approach used to estimate the demographic impacts of dam breaching involves examination of survival rates of juvenile downstream migrants over time. Direct effects

of the hydropower system should be evident as reduced survival after dam construction relative to survival rates prior to dam construction. Williams et al. (in review, North American Journal of Fisheries Management) conducted such analyses, estimating annual average survival probabilities of spring/summer chinook salmon from 1966 through 1999. They report estimates of survival ranging from 32-56% in the 1960s when only 4 dams were in place. After four additional dams were constructed from 1968 to 1975 survival dropped to 3-30% during the 1970s. After improvements to the hydropower system were put in place survival increased, and during the last 5 years, survival rates of juvenile migrants through eight mainstem dams ranged from 42-59% - a value equal to or greater than that in the 1960s. While survival through the hydropower system in the 1990s is apparently substantially greater than the 1970s, adult return rates have remained low. Consequently, direct mortality of downstream migrant fish through the hydropower system cannot be responsible for the continuing low adult return rates. However, this analysis does not eliminate the possibility that indirect effects of the hydropower system that are expressed as increased post-Bonneville mortality are important.

A different broad-brush tactic adopted by Regetz (in review, Conservation Biology) helps put dam passage in perspective. He contrasted the productivity of 22 different spring/summer chinook salmon stocks in the Columbia Basin that differed in habitat attributes and the number of dams they had to pass through before reaching the mouth of the Columbia River. Depending on the way recruit per spawner data was summarized, the number of dams (which ranged from 1 to 9) explained between 1% and 16% of the variation in recruits per spawner data (in no cases a was a statistically significant proportion of the variation explained by number of dams). This analyses indicates that it is not that easy to generate data that clearly and unequivocally point to dam breach as enough of an improvement to recover these stocks. Again, there is no question that dam breaching will provide some marked improvement, but data suggesting that it will be sufficient, by itself, are lacking.

V.A.7. The Bottom Line and Critical Uncertainties for Snake River Spring/Summer Chinook Salmon

Unless dam breaching increases survival below Bonneville Dam by upwards of 80-100 percent (combined with dramatic improvements in survival of upstream migrants), it seems unlikely that dam breaching, by itself, can recover Snake River Spring/Summer Chinook. It might seem surprising that dam breaching does not yield a dramatic and clear effect with minimal uncertainty, given the obvious impacts of dams. The reason for this is that the fish passage systems and barging of fish (most spring/summer chinook salmon are barged) are effective at getting fish to below Bonneville Dam (Williams et al. in review). In a sense, engineering has replaced nature for that portion of the salmon life cycle. There may be many ecological reasons to favor natural processes and natural rivers, but in terms of demographic accounting for spring/summer chinook salmon, those reasons have to be found outside the direct impacts of dams on fish migrating through the hydrosystem. The place to look for these effects may possibly be below Bonneville Dam, in terms of extra mortality caused by the

hydrosystem. Experimental approaches that manipulate the location of offshore releases, such as those of McNeil et al. (1991), would be an effective way of understanding and thereby minimizing this extra mortality resulting from transportation. Alternatively, it could be that the hydropower system alters survival during the first year because it impedes processes by which nutrients are replenished and the productivity of rearing habitats is maintained. If solutions outside the hydropower system are considered (habitat improvements, harvest reductions, hatchery modifications, predator control) a similar paucity of data relating management action to demographic improvements exists.

Given the inadequacy of dam breaching as the sole solution, there is a clear need to pay increasing attention to actions in the “other H’s”. Harvest is so modest for these stocks that major improvements cannot be gained by further reductions in harvest. This leaves hatcheries and habitat. The opportunities provided by habitat are made evident by Regetz’s recent study. Our preliminary analysis linking marine survival to hatchery releases in some years also illustrates a potential link between a specific management action and likely improvement to population growth rates. We urge that the emphasis now needs to be on feasibility studies linking specific management actions to likely demographic improvements. We know where to look for their greatest benefits from those links: in first year survival and in the early ocean - estuarine survival.

V.B. Snake River Fall Chinook Salmon

The data for fall chinook salmon are counts of fish passing the Lower Granite Dam on the Snake River (Table V-10).

Table V-10. Counts of Fall Chinook Adult Spawners (S) and Adult Recruits (R) (Peters et al. 1999). For this ESU adults are 3, 4, 5, & 6; jacks are age 2.

Year	S	R
1980	515	1236
1981	878	951
1982	1209	1201
1983	909	1054
1984	717	856
1985	1080	581
1986	1403	593
1987	1064	318
1988	702	778
1989	815	568
1990	273	233
1991	767	211
1992	674	
1993	883	
1994	448	
1995	226	
1996	964	
1997	1007	

As we did with spring/summer chinook, prior to matrix analyses we tested for evidence of density-dependence in fall chinook. For this ESU, the null hypothesis of density-independent population dynamics cannot be rejected (Table V-11). A second point worth noting about fall chinook salmon in the Snake River is that a very modest increase in lambda (only 2.5%) is required to mitigate the risk of extinction to below 5% in 100 years (Figure IV-11). Because the increase in lambda is modest, this suggests that many management options can substantially benefit fall chinook salmon in the Snake River.

Table V-11. Parametric bootstrap likelihood ratio test for density-dependence for Snake River fall chinook. a is the y-intercept and b is the slope from a linear regression of ln(recruits per spawner) vs. the density of spawners.

a	b	var	Test statistic	Critical value
0.5390	-0.00009	0.2951	2.5035	5.5071

V.B.1. Using a Matrix Model to Summarize Fall Chinook Demographic Rates and Opportunities for Recovery

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

	1	2	3	4	5	6
1	0	0	$(1-I)s_1b_3m_3/2$	$(1-I)s_1b_4m_4/2$	$(1-I)s_1m_5/2$	$(1-I)s_1m_6/2$
2	$(1-h_2)s_A$	0	0	0	0	0
3	0	$(1-h_3)s_A$	0	0	0	0
4	0	0	$(1-b_3)(1-h_4)s_A$	0	0	0
5	0	0	0	$(1-b_4)(1-h_5)s_A$	0	0
6	0	0	0	0	$(1-b_5)(1-h_6)s_A$	0

To derive parameter estimates for Snake River fall chinook, we used annual counts of natural-origin jacks and adults at Lower Granite Dam (1980-present) and age frequencies of spawners based on year-specific proportion at age calculated from Lyons Ferry Hatchery fall chinook CWTs (Peters et al. 1999; Table V-13). Mainstem harvest, ocean harvest, and *BontoBasin* conversion rates were also obtained from Peters et al. (1999). For harvest rates and survival during upstream migration, data from 1993-1996 were used, because there were reductions in harvest starting in 1993 under ESA management (Table V-13). Although there are potential problems involved with using data from hatchery fish, the best available information on age-specific fecundity and sex ratio at age come from fish at Lyons Ferry Hatchery (Mendel et al., 1996; Table V-13).

Table V-13. Age-Specific Parameters Used in Snake River Fall Chinook Analyses.

	2	3	4	5	6
Age frequency of females (f_x)	0	0.129	0.652	0.198	0.020
93-96 Ocean harvest rate (h_x)	0.0123	0.0465	0.1368	0.1838	0.1953
Eggs per female spawner (m_x)		2885	3133	3251	3251
Propensity to breed (b_x)	0	0.081	0.650	0.863	1.0
93-96 Mainstem adult harvest rate		0.174			
93-96 adult <i>BontoBasin</i> conversion rate		0.471			
s_1		0.0044167			

Age-specific propensity to breed (b_x) and survival during the first year (s_1) were calculated using methods similar to those described for spring/summer chinook. However, because fall chinook are ocean-type salmonids, s_1 includes everything beginning from egg hatching and including downstream migration, survival in the estuary and entry into the ocean environment. Because data regarding survival during downstream migration and the proportion of smolts transported are generally much poorer for fall chinook than for spring/summer chinook, no attempt was made to break s_1 down into all of its component pieces.

Table V-14. Parameterized Matrix and Population Growth Rate (I) for Fall Chinook Salmon.

	1	2	3	4	5	6	$I = 0.980$
1	0	0	0.182	1.573	2.170	2.513	
2	0.790	0	0	0	0	0	
3	0	0.763	0	0	0	0	
4	0	0	0.634	0	0	0	
5	0	0	0	0.229	0	0	
6	0	0	0	0	0.088	0	

As was done for the spring/summer chinook, the sensitivity of the matrix for fall chinook was evaluated in two ways: 1) elasticity analysis, and 2) numerical experiments investigating the percentage improvement associated with saving 1 out of 10 salmon that currently die at each stage. The elasticity results for fall chinook (Figure V-7) closely mirror those for spring/summer chinook salmon (Figure V-1). In particular, the most sensitive parameter is the survival of adults in the ocean, again because individuals at this stage have survived periods of high mortality and are close to the age of reproduction. Results of the saving 1 of 10 experiments for fall chinook (Figure V-8) are also quite similar to those for spring/summer chinook (Figure V-2). Specifically, reducing mortality during the first year of life produces the largest change in population growth rate (recall that for fall chinook, s_1 includes survival in the rearing habitat, downstream migration, and entry into the estuary and ocean environments). This result

can be largely attributed to the low estimated survival during the s_1 stage. Simply stated, because survival of s_1 fish is so low, saving 1 out of 10 fish that die at this stage would save many more fish than for any of the other stages.

It is more difficult to assess the potential benefits of dam breaching for Snake River Fall Chinook salmon because data regarding survival during downstream migration and the proportion of smolts transported are not as abundant. However, the majority of effects would likely occur in the s_1 stage, which includes both downstream migration and post-Bonneville survival in the estuarine environment (where latent effects of dams are likely to accrue). We examined the percent increase in lambda expected to result from a broad range of potential changes in s_1 survival. Again, an approximate 2.5 percent increase in lambda is expected to lower the probability of dropping to less than one spawner within 100 years to 5%. This level of improvement in lambda could be achieved with as little as a 15 % percent increase in s_1 (Figure V-9). Whether or not such a change in s_1 would actually occur under dam drawdown is unknown, but it seems highly likely.

In contrast to other proposed management actions, it is incontrovertible that harvest reductions will save fish from dying, and in fact save adult fish with far greater reproductive value than fry or smolts. Despite listing under the Endangered Species Act, harvest continues to be a significant source of risk to Snake River Fall Chinook salmon. Importantly, harvest reductions can take effect immediately, whereas dam breaching or habitat restoration will likely include time lags of 5-10 years before any effect is realized. The question then is, could harvest moratoriums attain the desired increase in annual population growth needed to mitigate the imminent risk of extinction of fall chinook salmon? Using a deterministic, stage-structured matrix model for fall chinook salmon, the required 2.5% increase in lambda is certainly achieved with a 75% reduction in mainstem harvest alone and may be accomplished with even a 50% reduction in mainstem harvest alone (Figure V-10).

Lastly, dam breaching would open up spawning habitat for fall chinook salmon, which, unlike spring/summer chinook, tend to spawn in the mainstem of the Snake River. However, expansion of populations to fill this habitat would not by itself reduce extinction risks; longer term increases in annual population growth rates above current levels would still be required. But the increased availability of habitat is a “benefit” not attainable through management of hatcheries or harvest actions.

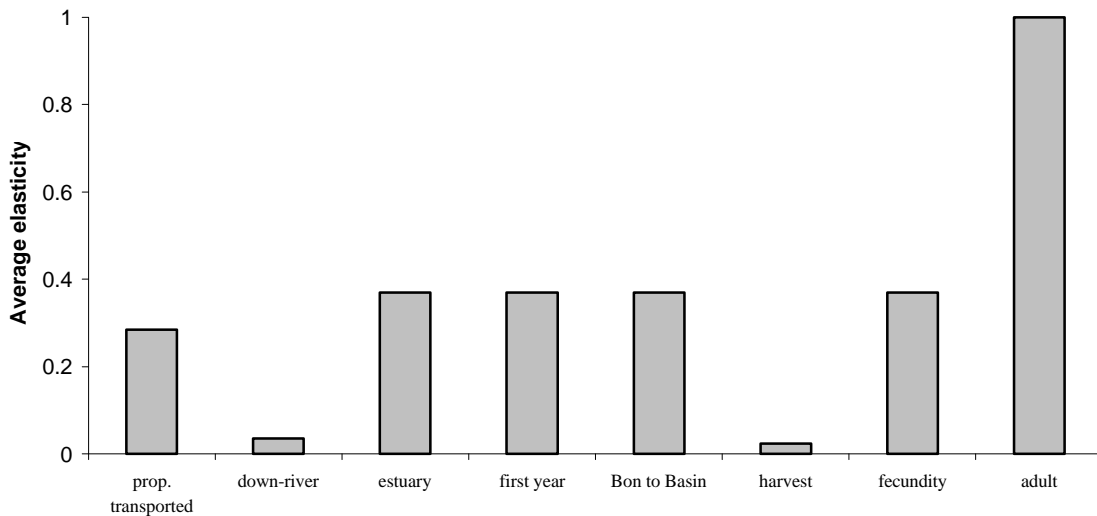
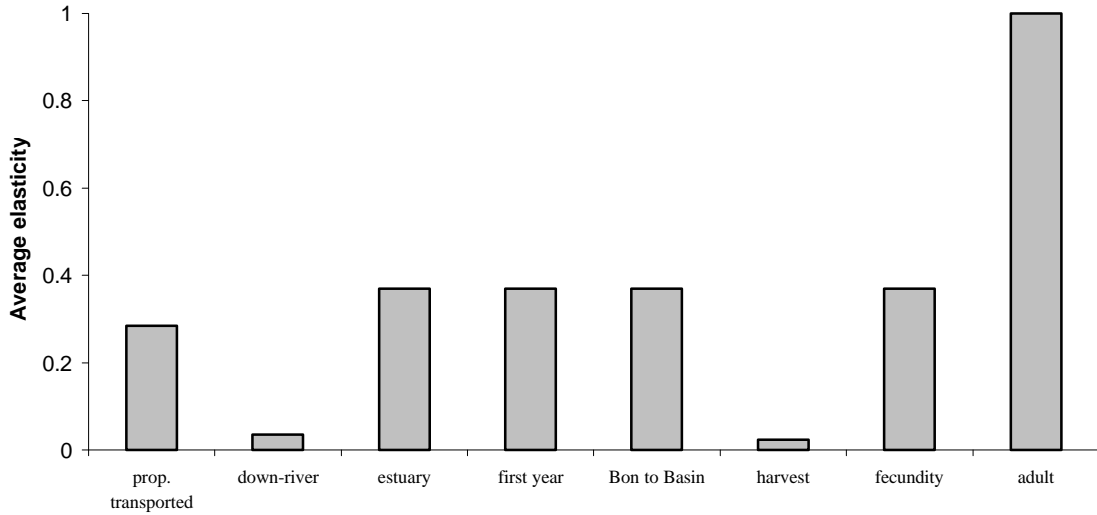


Figure V-1. Sensitivity of annual population growth to small changes in components of the Spring/Summer Chinook salmon demographic projection matrix. The top panel is derived from brood years 1980-1994, while the bottom panel is derived from brood years 1990-1994. Average elasticities, relative to the largest value, are shown.

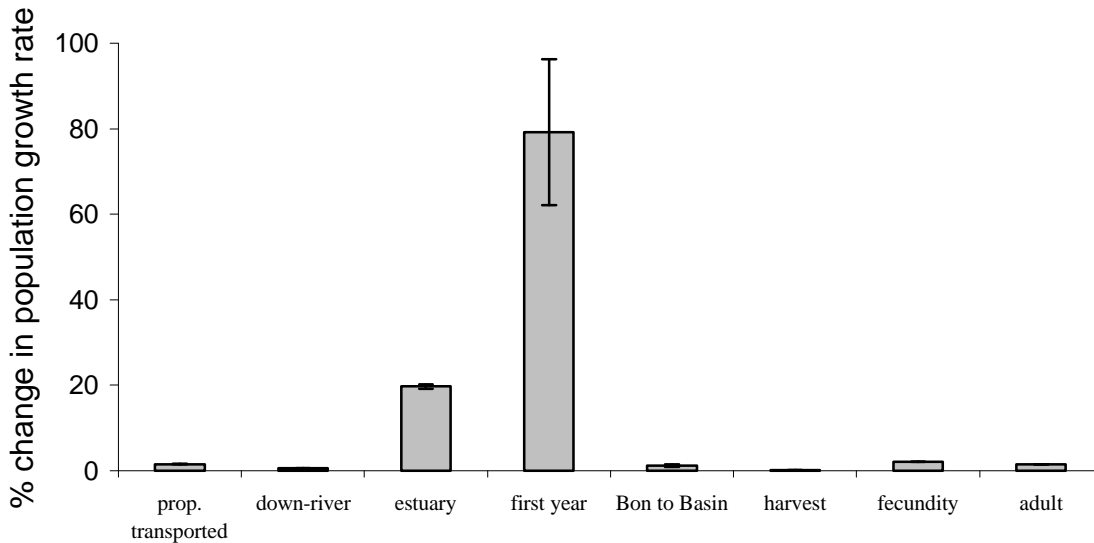
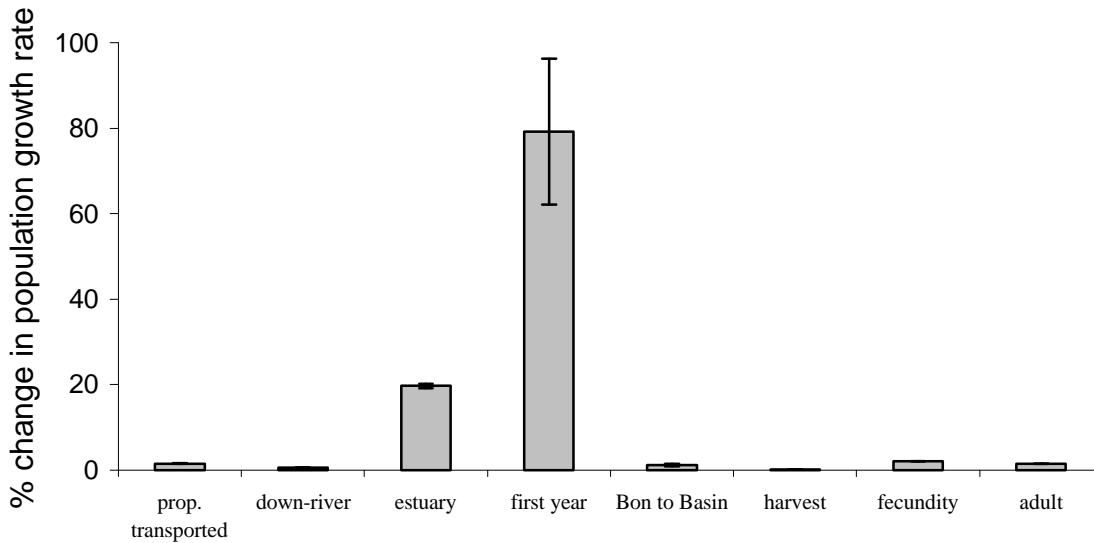


Figure V-2. Average increase in annual population growth with a 10 percent reduction in mortality during different life stages for Spring/Summer Chinook salmon assuming estuarine survival is 7%. A 10 % increase in fecundity, and a 10% increase in the proportion of smolts transported in barges were also simulated. The top panel is derived from brood years 1980-1994, while the bottom panel is derived from brood years 1990-1994. Error bars are ± 1 standard deviation.

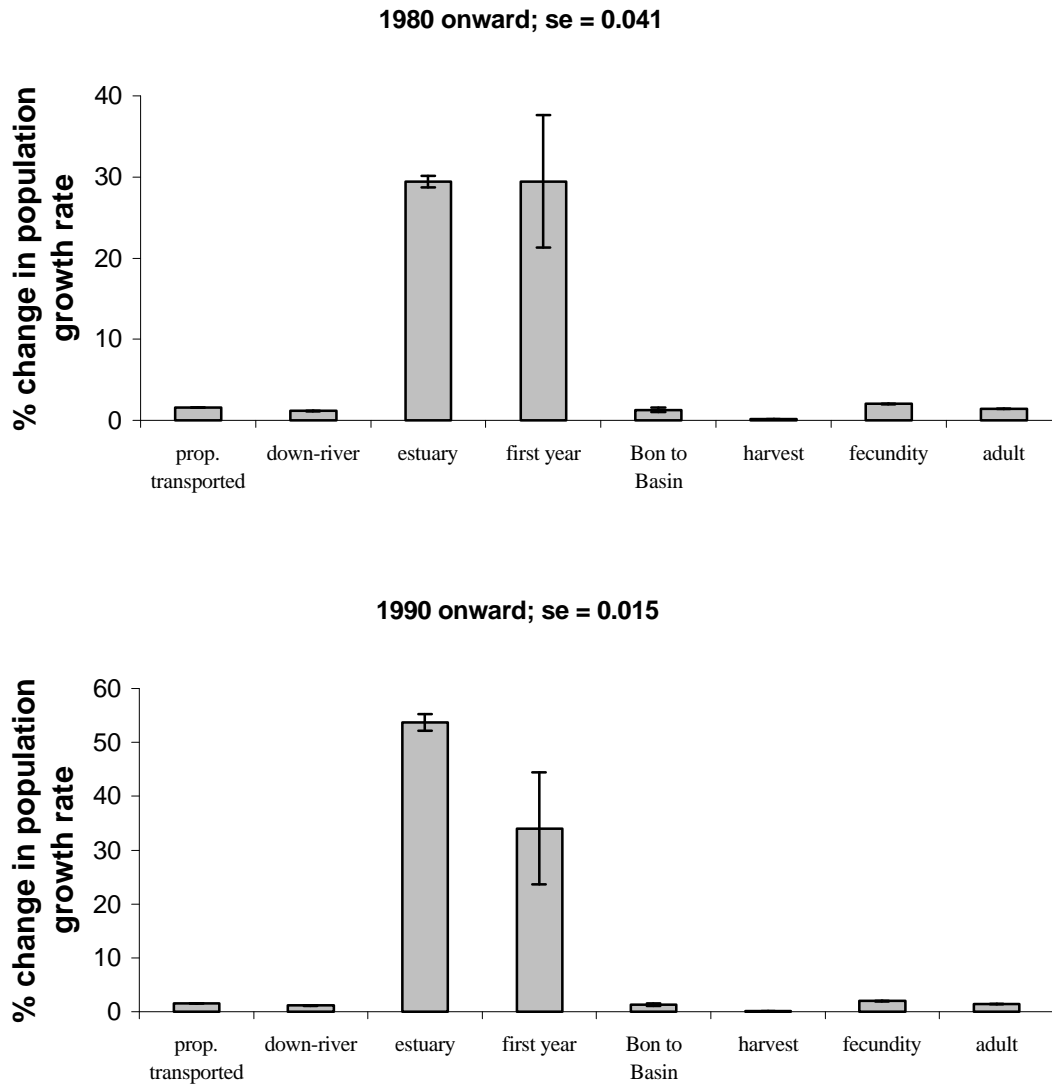


Figure V-3. Average increase in annual population growth with a 10% reduction in mortality during different life stages for Spring/Summer Chinook salmon. Estuarine survival (s_e) was calculated to produce the smolt-to-adult returns reported by PATH. The top panel is derived from brood years 1980-1994, while the bottom panel is derived from brood years 1990-1994. Error bars are ± 1 standard deviation.

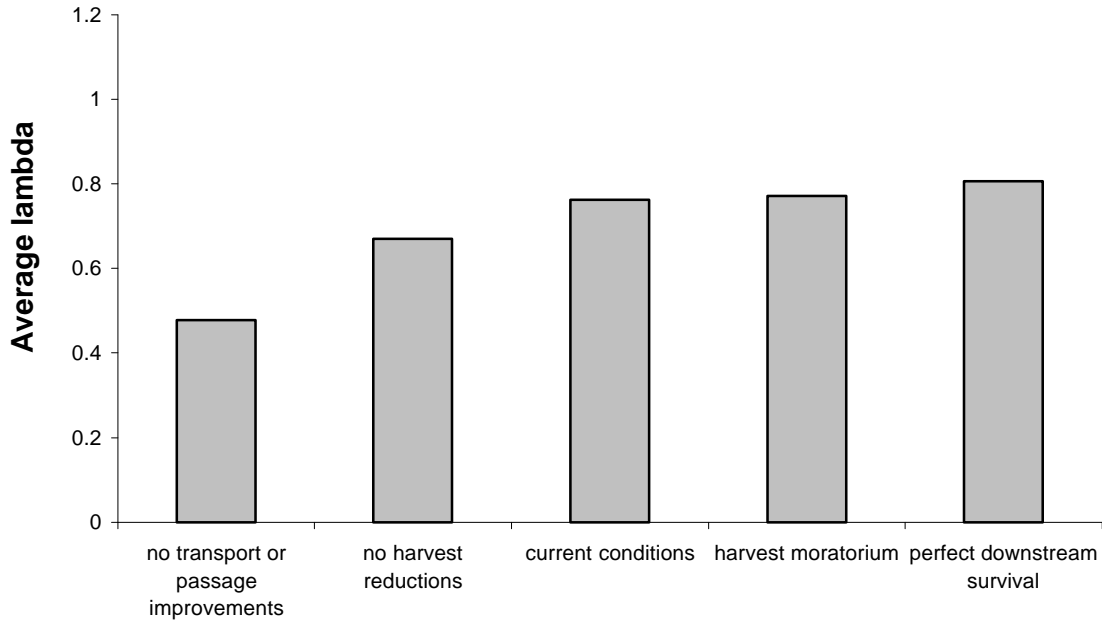


Figure V-4. Average rate of population growth for Snake River Spring/Summer Chinook salmon assuming different management scenarios. “No transport or passage improvements” equals current conditions, except $pt = 0$ and survivorship through the hydrosystem is set at rates observed between 1977-1979 (after dams had been constructed, but before improvements to bypass and transportation). “No harvest reductions” equals current conditions, except mainstem and subbasin harvest rates set to those seen between 1960 and 1970. “Harvest moratorium” equals current conditions, but with mainstem harvest entirely eliminated. “Perfect downstream survival” equals current conditions, except $pt = 0$ and $s_d = 1.0$.



Figure V-5. Possible Breaching Effects on Spring/Summer Chinook salmon Estimated Through Improved Estuarine and Ocean Survival in Conjunction with Improved Upstream Survival.

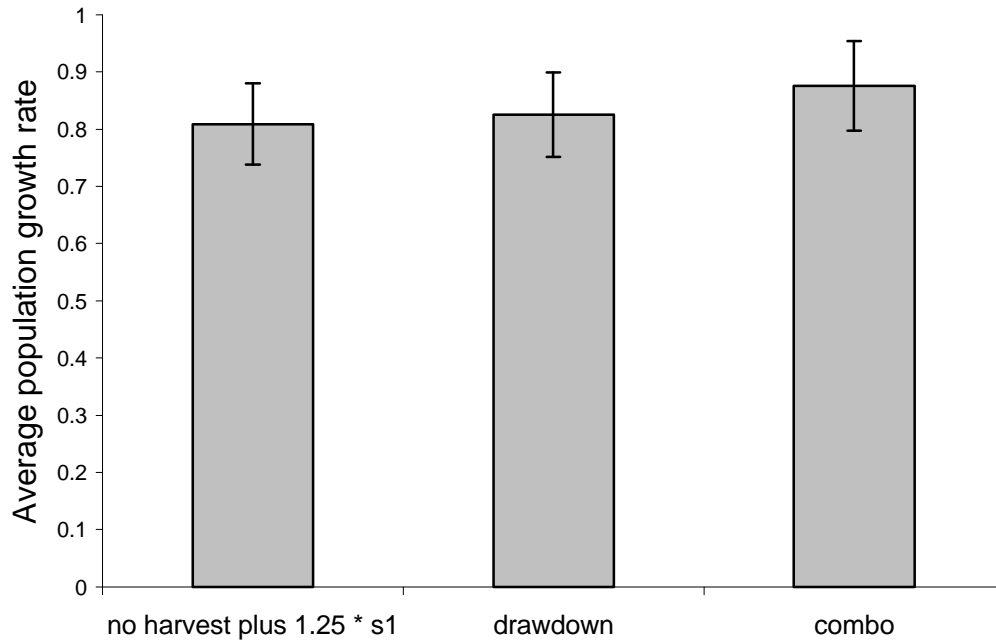


Figure V-6. Average Population Growth for Spring/Summer Chinook Salmon: Gauging the Effectiveness of Breaching Versus an Alternative Management Scenario. “No harvest plus $1.25 * s1$ ” represents current conditions without mainstem harvest but with a 10% increase in first year survivorship. Drawdown implies that dam breaching yields a 15% improvement in survival of adults migrating upstream, a smolt downstream migration survivorship of 62%, and a 60% increase in estuarine survivorship (roughly equal to $D = 0.5$). “Combo” is the combination of both scenarios.

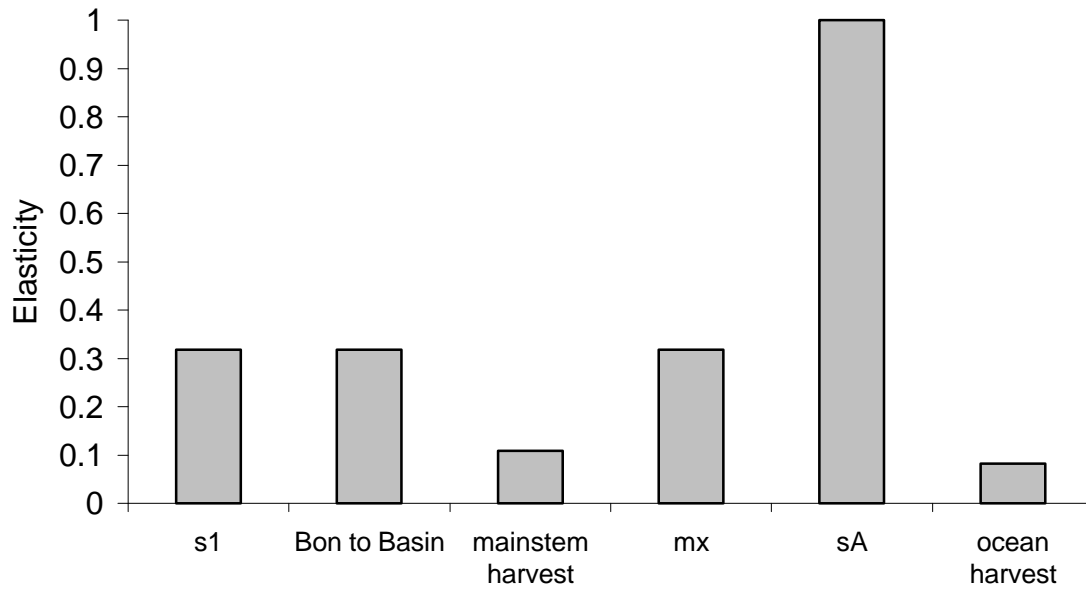


Figure V-7. Sensitivity of annual population growth to small changes in the components of the Fall Chinook salmon demographic projection matrix.

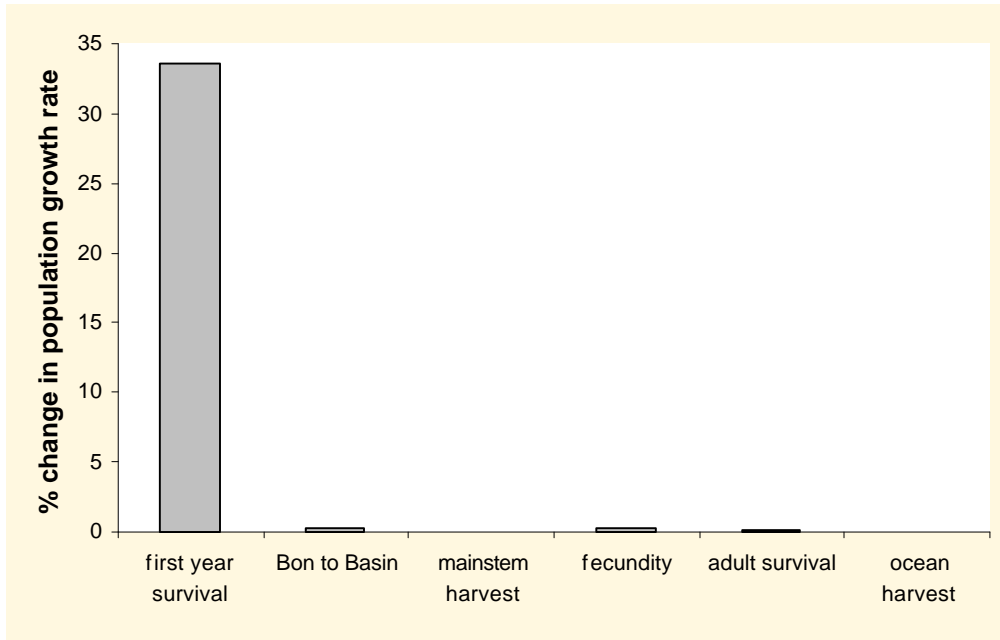


Figure 8. Improvements in Fall Chinook Salmon Annual Population Growth with 10 Percent Reductions in Mortality During Different Lifestages. A 10% increase in fecundity was also analyzed.

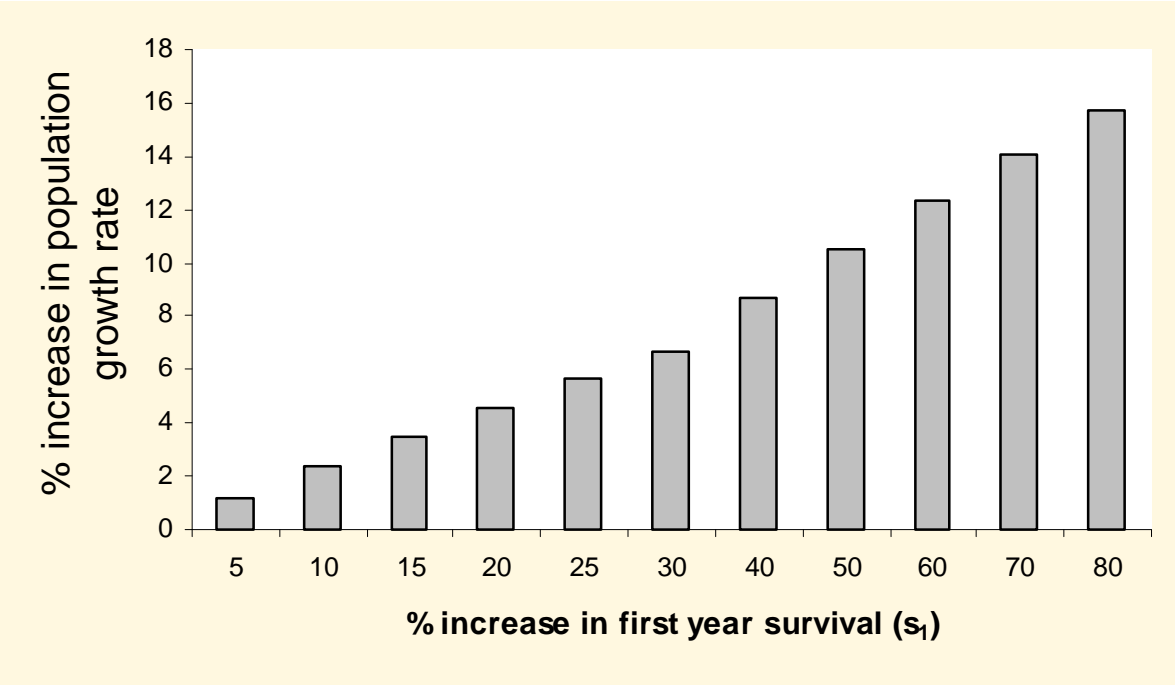


Figure 9. Increase in Fall Chinook Annual Population Growth with a Range Of Increases in First Year Survivorship.

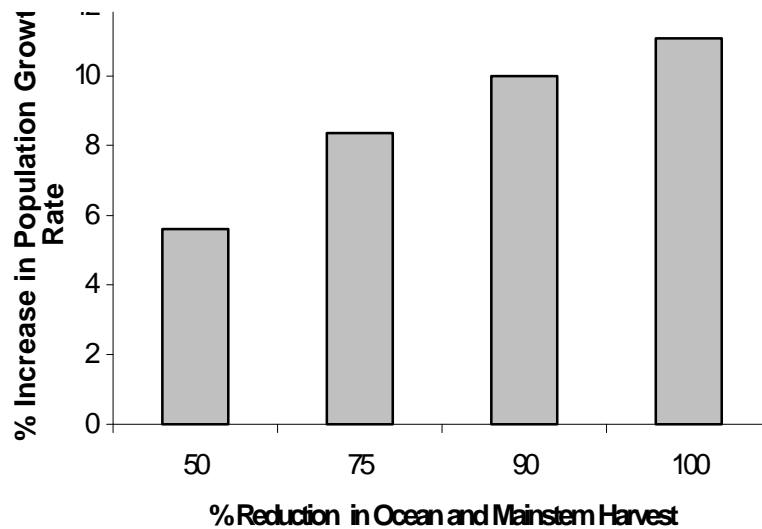
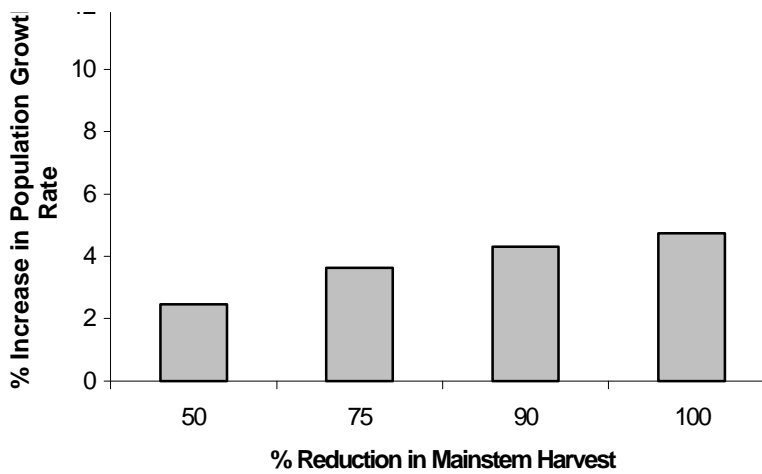
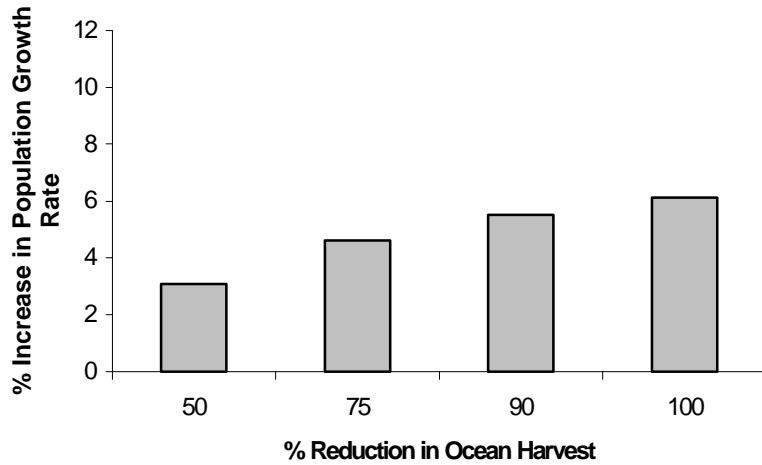


Figure 10. Results of a deterministic, stage-structured matrix model showing effects of harvest reduction on population growth.

VI. Detailed Demographic Models for Estimating Extinction Risks

Many published population viability analyses use detailed models of stage-specific survival, with the appeal that they are more biologically “realistic”. Unfortunately, even for the cases for which we have age-specific data (some chinook index stocks), we do NOT have estimates of variability in specific survival rates, or patterns of covariation among different demographic rates. As a compromise, however, we have developed a simulation model that is tailored to the age-structured salmonid life cycle, yet does not include detailed age-specific survival rates. This model is called SimSalmon, and is available on the CRI website. We do not apply it across all ESUs because the data are too sparse, but we are exploring its application for particular situations. The major problem with SimSalmon is that its estimates of environmental variability are likely to be greatly inflated by observation error, a problem less severe for the modified Dennis approach (as discussed above). However, unlike the simple modified Dennis approach discussed in III.A. SimSalmon can deal with much more complicated scenarios, assuming the data to support such analyses are forthcoming.

VI.A. SimSalmon

SimSalmon is a computer simulation program for estimating extinction risk of salmonids populations using counts of spawner abundance. The program is written in Java, has a graphical user interface and is publicly available through the NWFSC web site. The documentation that is downloadable with the application describes input file structure, parameter specification and information on interpreting results. The model underlying the program is based on the population projection equation

$$N_t = \sum_{i=1}^{\text{max. age}} N_{t-i} R_{t-i} A_i, \quad [6.1]$$

where N_x is the number of spawners at time x , R_x is the recruits per spawner at time x , A_i is the average proportion of spawners that are age i and $maxAge$ is the maximum age at maturation. The recruits per spawner, parameter, R_x , is a random variable that introduces environmental stochasticity into the model. In the basic model, R is assumed to be lognormally distributed and the two shape parameters of the distribution can be estimated from a time series of abundance that includes information on the age distribution of spawners. By the term “recruits”, we are referring to individuals that return to naturally spawn not some estimate of pre-harvest individuals. Recruits per spawner is assumed to be lognormally distributed because the probability of a spawner producing a given number of recruits can be considered as the product of a large number of probabilities affecting fecundity and survival. Such multiplicative processes tend to show a lognormal distribution. More complex scenarios involving temporal autocorrelation or trend in R can also be explored using SimSalmon, though confidently parameterizing such models from data can be challenging. The model is conceptually

related to the Dennis type models described below, but there are important differences that involve data requirements, interpretation of results and model flexibility.

The basic simulation protocol involves choosing an initial number of spawner for years one through $maxAge$. The number of spawners is then projected forward in time by looping through the equation above. This process is repeated 100's or 1,000's of times and the proportion of population trajectories that go extinct by a given period of time is reported as the "risk metric" or "probability of extinction" for that time horizon. The model utilizes a "quasi-extinction" threshold. If the average number of spawners/year drops below the threshold for $maxAge$ years in a row, then the population is considered

extinct. (That is if $\sum_{i=t}^{t+maxAge} n_i \leq threshold * maxAge$, the population is considered extinct at time $t + maxAge$.)

Density dependence is currently incorporated into the model by including an optional depensation threshold and/or ceiling on the total number of spawners. If the number of spawners drops below the depensation threshold, the recruitment, R , is zero for that year. Although recruitment for a given year may be zero, it does not mean that the population is necessarily extinct. There may still be individuals out in the ocean that will return to spawn the next year. A population is only considered extinct if it drops below the quasi-extinction threshold. If a ceiling is included in the simulation, when $N > K$, then N is set equal to K , where K is the carrying capacity or ceiling.

Confidence intervals can be determined for the extinction risk estimate by bootstrapping using a method analogous to that used in Dennis et al. (1991). It is easiest to work with $\ln(S_{t+1}/S_t)$ rather than R . Since R is lognormal, the distribution of $\ln(S_{t+1}/S_t)$ is normal. The mean, \mathbf{m} , and variance, \mathbf{s}^2 , for $\ln(S_{t+1}/S_t)$ are related to the mean, w , and variance, \mathbf{y}^2 , of the lognormally distributed R as follows,

$$\mathbf{w} = e^{\left(\frac{\mathbf{m} + \frac{\mathbf{s}^2}{2}}{2}\right)}, \quad \mathbf{y}^2 = (e^{\mathbf{m}})^2 * e^{\mathbf{s}^2} * (e^{\mathbf{s}^2} - 1) \quad [6.2]$$

The mean of the $\ln(S_{t+1}/S_t)$, \mathbf{m} , is distributed

$$N\left(\mathbf{m}, \frac{\mathbf{s}}{\sqrt{sampleSize}}\right). \quad [6.3]$$

The variance of $\ln(S_{t+1}/S_t)$, \mathbf{s}^2 , is distributed

$$Chi(sampleSize - 1) * \frac{\mathbf{s}^2}{(sampleSize - 1)}. \quad [6.4]$$

By taking repeated, independent draws from these distributions, then recalculating the extinction risk (after converting from $\ln(S_{t+1}/S_t)$ back to R), it is possible to obtain a distribution for the extinction risk. From the distribution of extinction risk, any desired confidence interval can be determined.

The model suffers from many of the same assumptions that afflict other PVA models.

1) Stationary Process –If parameters are to be estimated from data, the data must represent observations of a stationary process. Past observations must predict future behavior of the system.

2) Spawners/Spawner Lognormal – This is an important assumption to examine when estimating parameters from data. Most commercial statistics packages will test the fit of distributions, look for outlier data points and help test some of the stationarity assumptions. SimSalmon includes some diagnostics to test this assumption.

3) All variability and density dependence can be represented in the spawner/early juvenile life-stage.

4) Temporal Autocorrelation either does not exist or meets the assumptions of the SimSalmon autocorrelation analysis model (see SimSalmon documentation).

5) Single Population – The program models a single population. The “Calculate Spawners/Spawner” analysis will do some corrections involving data that included immigrant fish (e.g. hatchery fish). However, the projection used in the extinction risk calculation does not include immigration. This can be an especially troublesome assumption in estimating parameters from a very small populations because only a few immigrants can dramatically distort the recruits/spawner ratios.

6) The age structure is known perfectly. – Confidence intervals currently do not include uncertainty about the true age distribution of the population.

7) Density dependence can only be represented as a population ceiling and a depensitory threshold.

8) No Observation Error – In estimating parameters from data, all observations represent true abundance.

The key parameter in SimSalmon is the distribution of recruits/spawner (R/S). The equation used for calculating R/S from data is

$$(R/S)_t = \frac{\sum_{i=1}^{\max \text{ Age}} N_{t+i} A(i)_{t+i}}{N_t}, \quad [6.5]$$

where N_t is the number of spawners at time t and $A(i)_t$ is the fraction of age i individuals at time t . The presence of naturally spawning first generation hatchery fish can complicate estimates of natural productivity and R/S ratios. The equation for R/S if first generation hatchery fish are spawning is

$$(R/S)_{withHatchery} = \frac{\sum_{i=1}^{\max Age} (N_{t+i} - M_{t+i})A(i)_{t+i}}{N_t - gM_t}, \quad [6.6]$$

where N_t is the number of spawners at time t , M_t is the number of migrants into the population at time t (e.g. first generation hatchery fish) and $A(i)_t$ is the fraction of age i individuals at time t and g is the relative spawning success of hatchery fish as compared to the progeny of natural spawners. [6.6] gives the Natural Return Ratio (NRR) for the population. If $g = 1$, first generation hatchery fish are assumed just as effective at spawning as natural fish. If $g = 0$, first generation hatchery fish can not successful spawn in the wild. For many systems, it is suspected that g is greater than zero but less than one.

The average age structure is calculated as the total number of fish that spawn at a given age, divided by the total number of spawners in the entire data set. The sample size used for the bootstrap confidence interval estimation is the number of observations of R/S . Lambda, the average annual rate of increase (or decrease) in the population, is calculated from the data as

$$\lambda = e^{r + \frac{s^2}{2}}, \quad [6.7]$$

*where $r = \text{mean}(\ln(R/S))/\text{generation time}$,
with generation time = mean age at reproduction
and $s^2 = \text{var}(\ln(R/S))$.*

If there were no observation error in counting recruits or aging fish, then the extinction metrics for SimSalmon and the modified Dennis model should be almost identical. SimSalmon might appear to be a predictive simulation model, but we are reluctant to use it to forecast population dynamics. Like the modified Dennis approach, its best use is to estimate extinction risk, not to predict the number of fish in some future year.

VI.A.1. SimSalmon Results

So far, we have used SimSalmon to explored extinction risk in the Snake River Spring/Summer Chinook stocks. Analyses are under way for other ESUs, where appropriate data are available. The summary of results (Table IV-1), suggest, however, that SimSalmon does a poor job estimating population parameters. This is most evident by looking at the lambda values; all lambdas are greater than 1, and some are substantially greater than 1. The reason these annual rates of change are so high is that the estimates of variance are inflated by observation error (recall that lambda increases

as variance increases, see discussion in III.D.2.). In general, problems will plague any model that does not have some special method for getting around the hazard of sampling error exaggerating estimation of environmental variation. Hence, although the increased realism of models such as SimSalmon (or even more complicated lifecycle models) is appealing, such models will misconstrue extinction risks unless they can address the problem of sampling error.

Table VI-1: SimSalmon summary of recruits/spawner data for Snake River spring/summer chinook stocks. The sample size represents the number of “recruit per spawner” ratios available in the data or each stock.

Population	mean(ln(R/S))	var(ln(R/S))	lambda	sampleSize
Bear	-0.153	1.644	2.203	15
Imnaha	-0.346	0.698	1.312	15
Johnson	-0.157	0.873	1.496	15
Marsh	-0.642	3.052	4.018	15
Minam	0.073	2.27	3.164	15
Poverty	-0.174	0.927	1.53	15
Sulphur	-0.113	3.885	6.815	13

VII. Summary of Key Findings: The Ten Most Important Points.

- 1) Comparisons among and within ESUs requires a standardized analytical approach. From such an approach, it is possible to draw general conclusions using a common currency about the relative risks faced by stocks within and among ESUs. Standardization also allows an assessment of the relative level of demographic improvement needed for different ESUs to mitigate extinction risk.
- 2) ESU and populations within ESU vary greatly in their risk and their rate of decline (spanning more than an order of magnitude of variation).
- 3) While Snake River Chinook salmon are clearly at risk and worthy of concerted efforts, it is worth noting that these ESUs are not the most imperiled ESUs; more specifically, steelhead in general and stocks from the Upper Columbia (steelhead and chinook) are facing the greatest relative risks.
- 4) A major uncertainty in estimating these risk measures involves information (or documentation of information) regarding the presence and influence of hatchery fish on the spawning grounds. Data on the proportion of naturally spawning fish that are of hatchery origin and the reproductive contribution of these fish are largely lacking. In the absence of such data, estimates of true population trends of wild fish may be obscured.
- 5) One way of using standardized assessments of risk is to identify “targets for improvement.” Clearly the specification of such targets is a policy decision. However, the use of a common metric derived using the same methodology highlights those populations in the most dire circumstances. Moreover, matrix models provide the life history stage that management action should target.
- 6) Harvest reductions offer plausible way to reduce risk for the few ESUs that presently bear substantial harvest burdens. Harvest reductions improved λ by 1% to 30% depending on current harvest pressure and generation time. ESUs that may be particularly likely to benefit from harvest reductions include Lower Columbia Chinook, Upper Willamette Chinook and Snake River Fall Chinook.
- 7) However, for the majority of the ESUs, harvest reductions alone are unlikely to adequately mitigate risks. For these ESUs, there is a need to ensure that the methods and data for identifying needed actions in other H's are in place.
- 8) The most recent data for Snake River Spring/Summer Chinook salmon reveal that this ESU may be doing worse than was previously thought. It is now even less likely that dam breaching BY ITSELF will mitigate imminent risks faced by Snake River Spring/Summer Chinook salmon.

9) The data and metadata are generally inadequate for a more detailed analysis than we have conducted here. In particular, the quality of the data is rarely documented, and consequently sampling error is unknown. The design of monitoring and evaluation efforts is hindered by the absence of information about sampling error. Standardization and consistency are largely lacking in both methodology and documentation.

10) The emphasis now needs to shift from determining relative risk to feasibility studies that link specific management actions to likely demographic improvements.

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