

***PasRAS: A STOCHASTIC SIMULATION OF CHINOOK AND
SOCKEYE LIFE HISTORIES***

***PELTON ROUND BUTTE HYDROELECTRIC PROJECT
FERC NO. 2030***

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GLOSSARY

Term or expression	Symbol	Definition
“p” value	p	Probability that you are wrong in assuming a variable is significant at the α level shown.
α	α	Type I experimental error: probability that you are rejecting the null hypothesis when it is actually true: probability of erroneously deciding that there is a difference.
Age 0		Fish that are less than 1 year old.
Analysis of Variance	<i>ANOVA</i>	A statistical procedure for partitioning the variance of the response variables from experimental data into relative contributions of different stimulus variables.
Carrying capacity, sockeye	$K_{fry}(t)$	Maximum reservoir population. For sockeye, expressed as kg of fall age 0 <i>O. nerka</i> (sockeye and kokanee combined).
Coefficient of variation	<i>C.V.</i>	Standard deviation/mean, a measure widely used to facilitate comparing standard deviations between different sample sizes, because the standard deviations are normalized as a fraction of their respective means (Sokal et al. 1995).
Comma-separated variable	<i>CSV</i>	File format in which numbers are written to a text file, and are separated by commas. Excel and other analytical software programs can convert CSV files to numerical files.
Delayed density dependence		Mortality is assumed to be affected by population sizes in previous years, by mechanisms such as might result from previous years’ sockeye populations overgrazing the zooplankton in a lake, which then take time to recover and hence cause a food shortage for subsequent sockeye populations.
Density-dependent factor (such as mortality)		“Factors limiting the growth of a population that are dependent on the existing population density and that are generally effects of other species in the form of competition, predation or parasitism”

Term or expression	Symbol	Definition
		(Lawrence et al. 1998).
Deterministic function or variable		Function or variable that is assumed to be known, without random variation. In contrast to a stochastic variable, which is represented as some form of probability distribution, a deterministic variable is represented as a single number or algebraic function.
Efficacy		Effectiveness
Emigration		Departure, or attempted departure, en route towards the ocean, from rearing freshwater habitat: reservoir for sockeye, streams for chinook. Occurs when smoltification begins.
Estimated probability	\hat{p}	Probability used to define the binomial probability distribution.
Feedback mechanism		“General mechanism operative in many biological and biochemical processes, in which once a product or result of the process reaches a certain level it inhibits [negative feedback] or promotes [positive feedback] further reaction” (Lawrence 1998).
Fry		“Fish up to the time when the yolk sac has been absorbed” (Lagler 1956).
Fry-to-smolt survival	<i>Sparr</i>	Survival from fry stage up to emigration (smolt) stage.
Geometric mean	<i>Geomean</i>	Preferred method for representing the expected value when the variables are proportions (Ferguson and Takane 1989): the <i>n</i> th root of the product of <i>n</i> numbers.
Parr		Life stage between fry and smolt.
Recruits/Spawner	<i>R/S</i>	Ratio of number of surviving offspring that return to spawn, over a series of years, from a particular brood year.
Relative seeding		Current population of smolts expressed as a fraction of the maximum population of smolts

Term or expression	Symbol	Definition
		that a stream reach is assumed to be able to support.
Replication		In a Monte Carlo simulation like <i>PasRAS</i> , the entire simulation of, say, 50 years, is repeated—or replicated—many times. The concept is analogous to its usage in experimental design, where a replicate is “two or more physical samples, of the same size, that have been treated in an identical fashion” (Lawrence et al. 1998).
Smolt		Life stage of an anadromous fish when it is ready to emigrate to sea.
Smolt capacity, chinook	<i>N/A</i>	Maximum population. For chinook, expressed as number of smolts in any rearing reach.
Standard deviation	<i>s.d.</i>	Standard deviation (square root of variance).
Stochastic process or variable		“A process [or variable] in which there is an element of chance or randomness” (Sokal et al. 1995). Stochasticity or random variation in a simulation is what distinguishes it from a deterministic simulation. In contrast to a deterministic function or variable, which would be represented as a single number or algebraic function, a stochastic variable is represented as some kind of probability distribution.

ABSTRACT

PasRAS (Passage Risk Assessment Simulation) is a stochastic age-structured simulation of spring chinook and sockeye salmon life histories. The software was developed in order to help evaluate the feasibility of reintroducing spring chinook and sockeye salmon above the Pelton Round Butte (PRB) Hydroelectric Project on the Deschutes River in central Oregon. *PasRAS* is intended to be used as a “thinking” tool, for helping people reach agreement on the relative importance of risks, possible impacts of management activities, and prioritization of research and habitat improvement efforts. Although *PasRAS* is not intended to be used for predictive purposes, the fact that it produces results generally consistent with the longer-term patterns for spring chinook runs at Lower Granite Dam on the Snake (1975–1995), spring chinook runs above Sherars Falls on the Deschutes (1977–1998), sockeye runs to the Kenai (1969–1990), and sockeye runs to Lake Osoyoos on the Okanogan River (1960–1996), helps build confidence that the key variables are captured well enough to justify using it for the intended purposes.

The sensitivity analyses indicated that for spring chinook and sockeye, any of the parameters associated with ocean mortalities are critically important, both because the ocean is where much of the mortality occurs in the real world, and because of the high levels of stochasticity and unpredictability in the ocean. The most influential freshwater parameters for the sockeye simulation model were found to be:

1. Egg-to-fall-fry mortality
2. Juvenile collection efficacy
3. Spawner success
4. Reservoir habitat quality

The most influential freshwater parameters for chinook were found to be:

1. Tributary habitat quality
2. Juvenile collection efficacy and quality (as it affects juvenile mortality, smoltification and stress)
3. Columbia River dam mortalities
4. Mortalities suffered by adults returning up the Deschutes

INTRODUCTION

PasRAS (Passage Risk Assessment Simulation) is a stochastic risk assessment life history simulation of spring chinook and sockeye salmon. The software was developed in order to help evaluate the feasibility of reintroducing spring chinook and sockeye salmon above the Pelton Round Butte (PRB) Hydroelectric Project on the Deschutes River in central Oregon. *PasRAS* runs under Windows 95/98, as a stand-alone, user-friendly installable program written in Microsoft Visual Basic 5.0. Chinook and sockeye species models are both age-structured life cycle models that incorporate as many of the life history characteristics of the species as could be documented in the published and unpublished (gray) literature. The species models differ because of life history differences: the predominant life history type of spring chinook in the Deschutes rears in tributaries, whereas sockeye would be expected to rear primarily in Lake Billy Chinook (Figure 1).

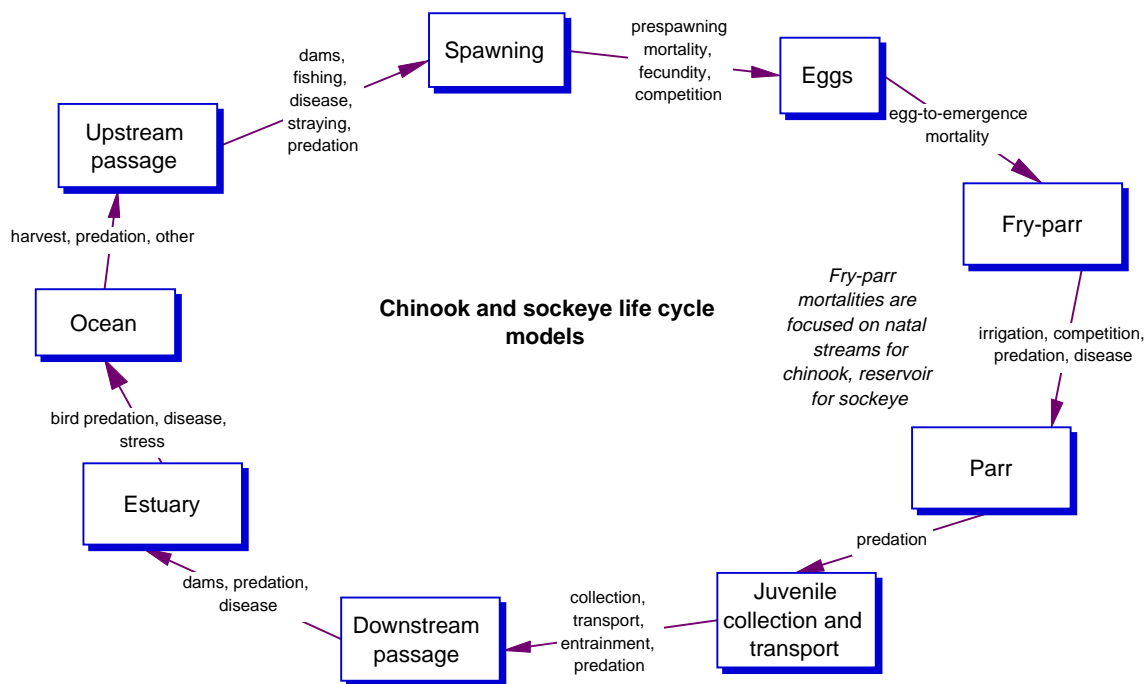


Figure 1. Life cycle schematic.

Risks to salmon survival at a sequence of life-cycle stages are modeled as a modified Markov chain of events. Monte Carlo methods are used to simulate uncertain events at each life-cycle stage (Burgman et al. 1993, Canner et al. 1992, Sonnenberg and Beck 1993). Most events are represented either as discrete (binomial) probability distributions, or normal probability distributions. The structural models on which the chinook and sockeye life cycles are based were developed by the Fisheries Technical Subcommittee (FTS, see Acknowledgements, p. iii) as part of their efforts to develop a decision structure for evaluating the feasibility of fish passage through the PRB project (Oosterhout 1998).

Using typical Monte Carlo simulation methods, *PasRAS* sets up and executes independent replicates of simulated life-cycle survival over many years. A typical replicate might simulate the life of the population over 100 years. In a Monte Carlo simulation like *PasRAS*, whenever the simulation needs a new value for some parameter, it samples from the appropriate statistical distribution (binomial, normal, exponential, uniform, etc.), which is defined by the appropriate parameters (mean, standard deviation (s.d.), coefficient of variation (C.V.), discrete probabilities, etc.) defined by the user. In order to achieve statistically meaningful results, it is typically necessary to run hundreds or thousands of replicates.

Each independent replicate of a Monte Carlo simulation is analogous to an independent sample drawn for a designed experiment. If properly conducted, a Monte Carlo simulation can be evaluated using statistical tools that could be applied to any designed experiment (Fahrig 1991). Just as with a designed experiment, the number of replicates required depends on the variance of the inputs (i.e., the range of uncertainty for each of the variables), and the sensitivity and power desired for the analysis. *PasRAS* produces results in several formats: probability density graphs for time to extinction and final population size; tables showing numbers of spawners, recruits, and recruits per spawner for each replication; graphs showing mean, median, minimum, and maximum population sizes year-by-year for each replication. Text files of key data are produced that can be opened in spreadsheet or statistical programs for further analysis.

PasRAS also allows the user to increase stochasticity and/or conduct multivariate sensitivity analyses by representing the probabilities themselves as ranges. For example, with the sensitivity analysis option turned on, instead of assuming a nominal 0.6 survival fraction at some life-cycle stage, the user can define survival as a range of, say, 0.4 to 0.8. Then, whenever *PasRAS* needs to use this parameter, it first randomly picks a value from the defined range, and then samples from the appropriate distribution using that value instead of the nominal value. To conduct a factorial experiment, the user can have *PasRAS* flip a coin rather than sample from the full range of a parameter: instead of sampling from a range of 0.4 to 0.8, *PasRAS* would use either 0.4 or 0.8, with a probability of 0.5 of using either. Sensitivity analysis can be conducted for up to nine parameters at a time. Even when the sensitivity analysis option is turned off,

PasRAS still uses Monte Carlo sampling tools in order to represent all variables by their appropriate statistical distributions.

This report describes how the two species models of *PasRAS* work, the validation process, and results of the sensitivity analyses and management scenario simulations.

THE *PasRAS* MODEL

Spawning Stage

Both the sockeye and the chinook species models initialize each replication with a starting population of returning spawners. The starting populations are sampled from probability distributions defined by the user. For both species models, straying rates are defined by tributary system. The rates of straying can be different for the three main tributary systems to Lake Billy Chinook (Crooked, Deschutes, and Metolius rivers), but straying itself occurs between reaches. To simulate straying, *PasRAS* first determines the number of strays that will not go back to their home reach by sampling from a binomial probability distribution for each reach (with \hat{p} = the tributary's stray rate), and adds them up into a total "pool" of potential strays. It then distributes this pool of strays proportionate to the fraction of the total population assumed to be in each reach. Thus, the fraction of spawners straying away from a reach is approximately \hat{p} , and they are redistributed proportionately to the expected fraction of the total population assumed to spawn in each reach. Strays could also end up in, or come from, another watershed altogether, of course, but this effect has not been added to *PasRAS* as yet. Because it is not known what drives straying, this method is a rough approach intended primarily to introduce a small amount of stochasticity related to straying, rather than to faithfully reproduce metapopulation dynamics.

For both the sockeye and chinook species models, parameters needed at the spawning stage are resampled from the appropriate distribution every time they are needed, for each reach and/or system, for every generation and every replication:

- Male-to-female ratios are defined as a range rather than a single ratio. The user defines a uniform distribution characterized by the minimum and maximum fraction female expected. For each reach, the simulation calculates the expected fraction of females by first sampling from this uniform distribution. Next, it samples from a binomial distribution using this value of \hat{p} .
- Pre-spawning mortality is defined by the user, and the simulation uses this fraction to sample from a binomial distribution with \hat{p} = pre-spawning mortality.

- Baseline fecundity is defined by the user as a normal distribution with a mean of \bar{x} and s.d. of s . For the first few generations, the number of eggs per female is recalculated each generation, each reach, by sampling from this distribution. Age structuring is assumed to be established after the simulation has run the same number of years as there are year classes. The number of year classes is defined by the user, but for sockeye and spring chinook, it would typically be three. Thus, establishing age structuring in the model would typically take three years. Once age structuring is established, fecundity is defined as a function of spawner size. Although growth rate is a function of ocean conditions, at this time in *PasRAS*, size in the ocean is defined in terms of age alone. Fecundity F is defined in terms of length L as :

Sockeye (Foerster 1968): $F = 86.8 L - 1952$

Chinook (Healey 1991): $F = 0.00195 L^{2.234}$

Length is defined by the user according to time spent in the ocean. Several fecundity functions for chinook were considered (Healey 1991, Nicholas and Hankin 1988). The one chosen, which is from the Quinsam River, was chosen because it produced average mean fecundities consistent with available data from the Columbia (Mullan et al. 1992, Fagan 1998, Lindsay et al. 1989).

At the end of the spawning stage, the population for each reach is assumed to be the calculated fecundity times the number of females, which has been reduced by prespawning mortality and straying out, and increased by straying in.

Egg-to-Emergence Stage

Egg-to-emergence stage mortalities for habitat quality rated Good, Fair, and Poor are defined for normal years as well as for catastrophic event years. Increased mortality due to a catastrophic event such as a flood or drought is handled the same way for both species models. The user defines the catastrophic event frequency in terms of events per 100 years. *PasRAS* assumes that catastrophic events exhibit a Poisson distribution. For a Poisson distribution, the number of years before a first event occurs, or time between events, can be simulated by sampling from a geometric distribution (Burgman et al. 1993, p. 280):

$$\text{Time to event (years)} = \ln U / \ln (1-p)$$

where:

U = random number between 0 and 1

p = probability of occurring (e.g., 4 flood years out of 100 would yield $p = 0.04$)

In the simulation, “time to event” is re-calculated at the beginning of each replication and then again any time a catastrophic event occurs. Just as in nature, it is thus possible for catastrophic events to occur two generations in a row in one replication, or never in another replication, even if the assumed frequency is four events per century. The catastrophic event option can also be used to increase stochasticity of the egg-to-emergence stage, by defining a high event frequency with only moderate impact.

An additional mortality intended to account for genetic effects at small population sizes is also included at this stage. The user defines a range of mortalities in terms of the effective population size. The method for doing this is similar to that used by Nickelson for coho (Nickelson and Lawson 1998), which was based on work by Lynch (Lynch 1996). Nickelson assumed the effective population to be 0.6 times the actual spawner population for coho, arguing that although the standard effective fraction is approximately 0.2, if there is genetic interaction among three successive broods, 0.6 should be used instead. The user can define the fraction as desired. As it turns out, the effect of which assumption is used is undetectable, because the effect is small compared to other mortalities, though it may have some effect on extinction probabilities at small population sizes.

Fry to Smolt Stage: Chinook Version

For chinook, survival from emergence to emigration is determined by density-dependent parr survival curves. Four curves can be defined: one each for Good, Fair, and Poor rearing habitat quality, and a fourth for a chinook life history that rears in the reservoir instead of in the streams (“ocean-type”). The user does not have to define these curves differently, and does not have to include the reservoir-rearing life history, but *PasRAS* makes it easy to do so, in a somewhat crude way, if desired¹.

¹ A problem with the reservoir-rearing chinook life history stage as currently implemented is that *PasRAS* ignores the fact that ocean-type spring chinook do not spend the winter in freshwater but instead emigrate out in the fall. This means that the fraction assumed to be ocean-type will be treated as though they are the same age as if they had

Parr survival depends on relative seeding (Nickelson and Lawson 1998):

$$S_{parr} = K * F^{-r}$$

Where:

K is the survival rate at full seeding

F is the current population expressed as a fraction of full seeding

r is the slope of the declining exponential equation at low seeding.

r represents the degree to which the population exhibits density dependence. If r is zero, the curve is a horizontal line, and population is not density dependent. Some example curves are shown in Figure 2.

Parr survival curves: $S = K * F^{-r}$

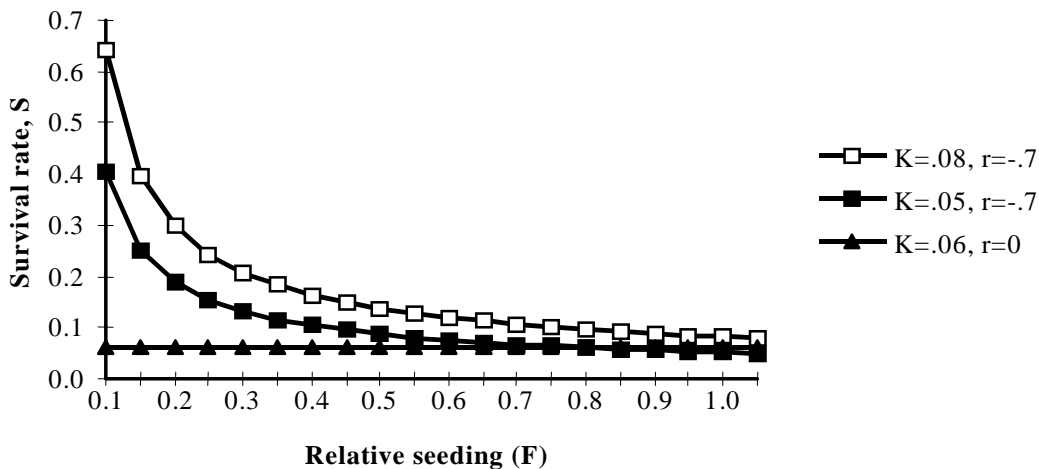


Figure 2. Chinook parr survival curves.

stayed as long as the stream-type life history, when in reality they will tend to be smaller smolts and probably subject to higher mortalities.

The maximum seeding level that defines the parr survival curves is defined in the reach characteristics table (Figure 3). In the chinook version, each reach is defined as Good, Fair or Poor in spawning and rearing habitat quality, and to be part of the Metolius, Deschutes, or Crooked River tributaries to Lake Billy Chinook. The user can define reach characteristics for as many reaches as desired.

Initial seeding Click on cells to see HELP message in status bar.

Number of spawning reaches = 3

Reach #	Name	Spawning HQ	Rearing HQ	System	Max smolt capacity	Initial spawners
0	Metolius	3	1	1	333426	3500
1	Deschutes	2	2	2	222284	3421
2	Crooked	1	3	3	111142	584

Read CSV file Set stray rates Done

7:14 AM

Figure 3. Spawning reach definition defined table for chinook, showing numbers only as examples (different values were used for different scenarios. Numbers shown here are for illustration only).

Relative seeding is calculated as follows.

Since the parr survival curve constant, K , is the survival rate at full seeding, the maximum parr at full seeding would be the smolt capacity at maximum seeding divided by the parr survival rate K at full seeding. For any reach, then, relative seeding at the parr stage is the actual number of parr (before mortality is calculated) divided by the maximum number of parr at full seeding. This approach is similar to that used by Nickelson and Lawson (Nickelson and Lawson 1998) in their coho salmon model. The difference is that they assumed all populations had the same survival curve, whereas in *PasRAS*, the user can assign different curves for different habitat qualities, including curves that are flat (no density dependence). In addition, because at very low seeding, for high K values, survival can exceed 100%, *PasRAS* allows the user to cap the survival at some value.

Some variation is introduced in S_{parr} by adding stochasticity to density-dependent parr survival as follows: the user can define a normal distribution function where the mean = nominal survival and the s.d. is some constant fraction f of nominal survival (i.e., C.V. = f). With this approach, variance increases with lower population sizes because with the ratio of the mean to the s.d. constant, if the mean gets smaller, then the s.d. gets larger.

In the simulation, the parr survival rate is re-calculated every generation for every reach, and the reach population is decremented by sampling from a binomial distribution with $\hat{p} = 1 - S_{parr}$.

For the chinook life history that rears in the natal streams (“stream-type”), mortality in the reservoir is neglected because the fish are assumed to spend little time there. For the life history that rears in the reservoir (“ocean-type”), reservoir mortality is accounted for by the reservoir-rearing parr survival curve.

Fry to Smolt Stage: Sockeye Version

Sockeye fry are not assumed to stay in their natal streams very long, but instead to move relatively quickly down to the reservoir. Mortality in the tributaries, from emergence to spring fry stage, includes mortalities in the tributaries due to predation and unscreened irrigation diversions.

Unlike chinook, which are assumed to spend very little time in the reservoir, sockeye spend a year or two there. Also unlike spring chinook, which would be the only significant *O. tshawytscha* in the lake, sockeye would interact with conspecific, resident kokanee populations. Sockeye and kokanee fry both feed on various species of zooplankton. Zooplankton abundance is not easy to model for several reasons, but in particular because of interactions among sockeye, kokanee, and zooplankton, across multiple years. Bull trout predation is a major source of mortality for young kokanee, as it presumably would be for sockeye.

A subgroup of the FTS was formed to investigate how these interactions might be modeled. That group consists of:

Phil Mundy (independent consultant, Fisheries and Aquatic Sciences)

Don Ratliff [fisheries biologist, Portland General Electric (PGE)]

Steve Thiesfeld [fisheries biologist, Oregon Department of Fish and Wildlife (ODFW)];
later replaced by Chris Kern (fisheries biologist, ODFW)

Mike Gauvin [fisheries biologist, Confederated Tribes of the Warm Springs Reservation of Oregon (CTWS)]

The sockeye modeling work group developed a conceptual model for the interactions among sockeye, kokanee, bull trout, and zooplankton, across multiple years (Figure 4).

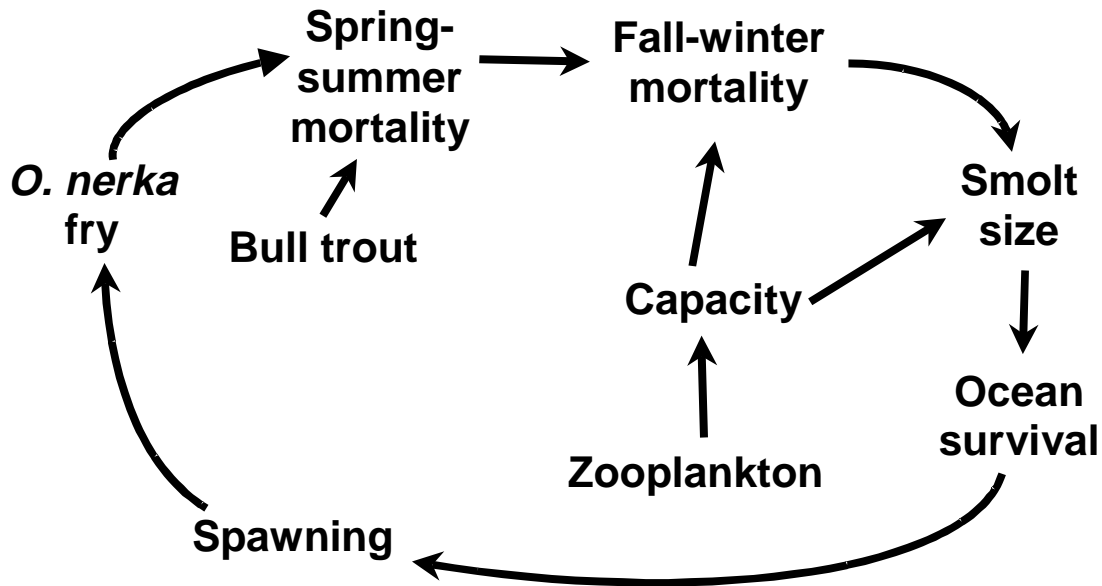


Figure 4. Sockeye fry survival model.

This model was converted to computer code, which *PasRAS* uses first to estimate each year's reservoir carrying capacity $K_{fry}(t)$ for *O. nerka* (sockeye and kokanee combined), and from that, to estimate fry mortalities. A detailed mathematical description of this approach is provided in Appendix A. Essentially, this part of the model works as follows:

Sockeye fry arrive at the reservoir in the spring. Mortality from arrival through emigration is modeled in two stages: (1) a spring-summer stage, in which bull trout predation is the predominant mortality, followed by (2) a more strongly density-dependent over-winter mortality. Bull trout predation is modeled as a density-independent, constant predation level (in numbers of fry) subject to a fractional limit at low populations (Mundy 1998–1999). Density-dependent over-winter survival is defined similarly to the way chinook parr survival is defined, by exponentially declining curves (Figure 5) that can be defined anywhere along a continuum from flat (no density dependence) to strongly density dependent (very steep slope).

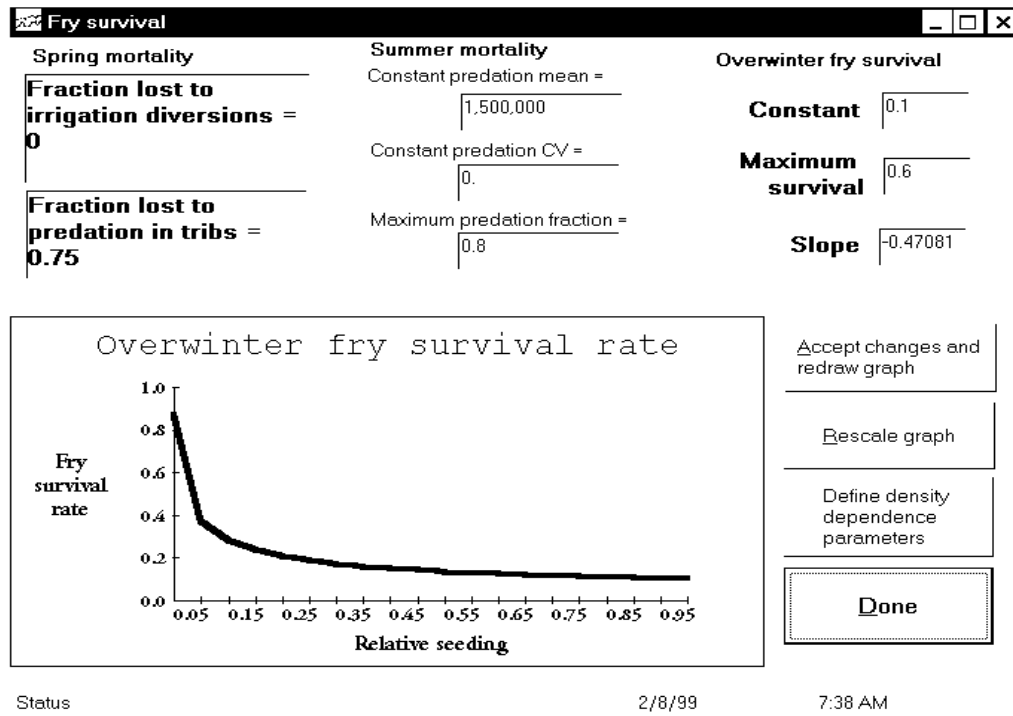


Figure 5. Sockeye fry survival screen. Parameter settings shown are examples only.

Density-dependent mortality is a function of *O. nerka* fall fry population size as a fraction of that year's carrying capacity, which is analyzed in kilograms rather than numbers of fish. The *O. nerka* population consists of the current year's sockeye and fry. Kokanee are modeled as varying randomly within a range defined by the user. Obviously, this is a significant oversimplification, but a more realistic model of kokanee population dynamics has not yet been developed. Similarly, reservoir carrying capacity for *O. nerka* is modeled by sampling from a uniform distribution defined by the user.

PasRAS provides flexibility in that any of these assumptions can be ignored: fry mortality can be defined to be a constant, non-density dependent fraction from emergence through emigration, for example.

The slope of the fry survival curve shown in Figure 5 indicates that when the *O. nerka* population is small relative to the carrying capacity (e.g., "Relative seeding" = 0.05), then nominal over-winter mortality M_{fry} would be relatively low (e.g., survival \approx 0.4). If the population is large relative to the carrying capacity, then M_{fry} would be relatively high (e.g., survival = 0.1, the constant in the fry survival curve). If the slope is zero, then nominal mortality would be the same for all population sizes. Mortality is implemented by sampling from a binomial distribution where $\hat{p} = M_{fry}$. An upper limit to survival can be defined so that when relative seeding is very low, survivals 'do not exceed that value.

Juvenile Collection

For both chinook and sockeye, mortalities associated with juvenile collection efficacy are assumed to consist of three components: the fraction of juveniles that can be attracted to the forebay (f_a); the fraction of juveniles attracted that escape entrainment ($1-f_e$); the fraction of juveniles that escape entrainment and are successfully collected f_c , and the fraction of those that are successfully collected who are also successfully transported (f_t). The net efficacy is:

$$\text{Efficacy} = f_a * (1-f_e) * f_c * f_t$$

In the simulation, the populations in various reaches are decremented by sampling from a binomial distribution with $\hat{p} = \text{Efficacy}$.

Downstream Migration Stage (Juveniles)

Downstream migration is modeled the same way for chinook and sockeye. Mortalities are defined for downstream passage through the Deschutes, as well as for the downstream passage through the Columbia. Juveniles have to negotiate two dams on the Columbia: Bonneville Dam and The Dalles Dam. There is an additional threat in the Columbia estuary due to bird predation, which has become more serious in recent years.

Researchers are also concerned about increased mortality due to the juveniles being unwilling to enter saltwater until they are fully smolted (Dickhoff et al. 1995; Schreck 1998–1999). In natural systems without dams, biology determines when and how quickly fish move where, but when fish need to be transported past dams, some of them may not actually be ready to move out of the reservoir. As a result, they will tend to wait in the estuary until the smolting process is complete. Researchers at Oregon State University (OSU) have conducted extensive research into this problem (sponsored by the Army Corps of Engineers), and Dr. Carl Schreck contributed the estuary mortality definitions used in *PasRAS*.

In the estuary simulation, the population can be partitioned into nine possible stress/disease states: (1) unstressed, healthy; (2) unstressed, infected but not sick; (3) unstressed, sick; (4) moderately stressed, healthy; (5) moderately stressed, infected but not sick; (6) moderately stressed, sick; (7) stressed, healthy; (8) stressed, infected but not sick; and (9) stressed, sick. These fish are subjected to two kinds of mortality, depending on how sick and/or stressed they are: disease mortality, and “other” mortality. In addition, the population is partitioned into fractions that are fully smolted, moderately smolted, and unsmolted. These fish are subjected to two kinds of mortality: predation mortality, and “other” mortality. All these estuary mortalities are defined by the user, and implemented by sampling from binomial distributions.

Disease

In addition to generic disease impacts included in the estuary stage, impacts from four diseases can be included explicitly in the juvenile, smolt, and adult stages (for both the chinook and sockeye species models). The four diseases are: bacterial kidney disease (*Renibacterium salmoninarum* or BKD), Infectious hematopoietic necrosis (IHN), furunculosis (*Aeromonas salmonicida*), and ceratomyxosis (*Ceratomyxa shasta*). Mortality estimates used in the rest of the simulation undoubtedly include disease mortality, but very little information is available about how much background mortality is due to disease in the wild. Thus, ordinary disease impacts are already accounted for by mortality estimates in the various stages, and the user must take care not to double-count disease impacts.

One of the major risks of reintroducing salmon species above the PRB project is believed to be the possibility of introducing pathogens to which existing species may not be resistant. These additional disease impact options included in *PasRAS* are intended to represent possible disaster scenarios or to conduct sensitivity analyses, rather than to simulate ordinary life cycles. These disease impacts are a function of probability of exposure to each disease at a particular life stage, and extent of mortality if exposure occurs. Mortality is assumed to occur at any stage during or following the exposure stage. These probability estimates are represented not by discrete numbers, but rather as ranges of numbers defining minimum and maximum expected exposure probabilities, and minimum and maximum expected mortality rates. For each iteration, and each generation, the simulation determines disease impacts by sampling from uniform distributions defined by these ranges. In this way, disease can introduce considerable stochasticity to the simulation, presumably in a way that is similar to stochasticity in the wild.

Ocean Stage

The user defines the number of years adults can spend in the ocean. For spring chinook and sockeye three years are assumed, but up to five years can be defined. The user defines a discrete distribution of maturation rates (in terms of years in the sea). On arrival, the adults from each brood year are stochastically distributed into groups of fish classified by maturation rate. Each year class is subjected to “ocean mortality” decrements every year, so that the longer a particular fish stays in the ocean, the more chances it has to die. “Ocean mortality” is any mortality not due to fishing.

Different “ocean mortality” probabilities can be defined for each year a fish spends in the ocean. For example, the user can define “ocean mortality” so that most of the mortality occurs very early in the time a salmon spends in the ocean, and the probability of dying declines as the fish

grows and becomes less susceptible to predation. The time distribution of total marine mortality is thus dependent on the amount of time spent in the ocean. Delayed mortalities in the ocean due to stress or disease caused by downstream passage are represented in the estuary step.

For sockeye, the size of emigrants is determined by reservoir population density, as described above (see also Appendix A). In the chinook version, the FTS decided to neglect the effect of emigrant size on ocean mortality. Instead, for chinook, the user defines one expected mortality for each year in the ocean, along with a C.V. for each year. In the sockeye version, rather than defining one mortality for each year in the ocean, the user defines one mortality for “big smolts” and one for “small smolts” for each year in the ocean.

Emigrant populations that are small relative to the current carrying capacity are assumed to produce relatively big smolts, and thus potentially smaller ocean mortalities. *PasRAS* interpolates along a linear function in order to estimate the actual annual mortality to be applied for each year class of sockeye. If the user prefers to ignore this assumption, then mortalities for “big smolts” and “small smolts” do not have to be different.

For example, Figure 6 shows a linear ocean mortality curve that would be used for the first year in the ocean, if the user defined “big smolt” mortality to be 0.2 (20%) for the first year, and “small smolt” mortality to be 0.8 (80%) for the first year. Different sockeye year classes can have different mortalities assumed, because mortalities throughout their time in the ocean depend on the size of a year class when they first arrived in the ocean; in other words, growth is proportional to initial size. If the user only wants to define the first year mortality as being a function of smolt size, *PasRAS* allows them to do so.

Ocean mortality for the first year a chinook or sockeye spends in the ocean can also be represented by using the Aleutian Low Pressure Index (ALPI). Ocean mortalities for some salmonid species may be correlated with the ALPI (Beamish and Bouillon 1993), which is a cyclic variation in ocean conditions (Figure 7). If the ALPI is selected as the way of representing first year ocean mortality, the user can select any year as a starting point in order to indicate where the user wants *PasRAS* to assume that the pattern begins. The user has to define annual mortalities for each of the four stepped levels shown in Figure 7.

If the user selects the “Use ALPI” option, *PasRAS* uses these mortalities for first-year survival instead of the nominal first-year survival. If ocean conditions are assumed to be improving it would correspond to starting at about year 5. If ocean conditions are assumed to be getting worse, it would correspond to starting at about year 76. The ALPI cycle repeats itself in the simulation as required.

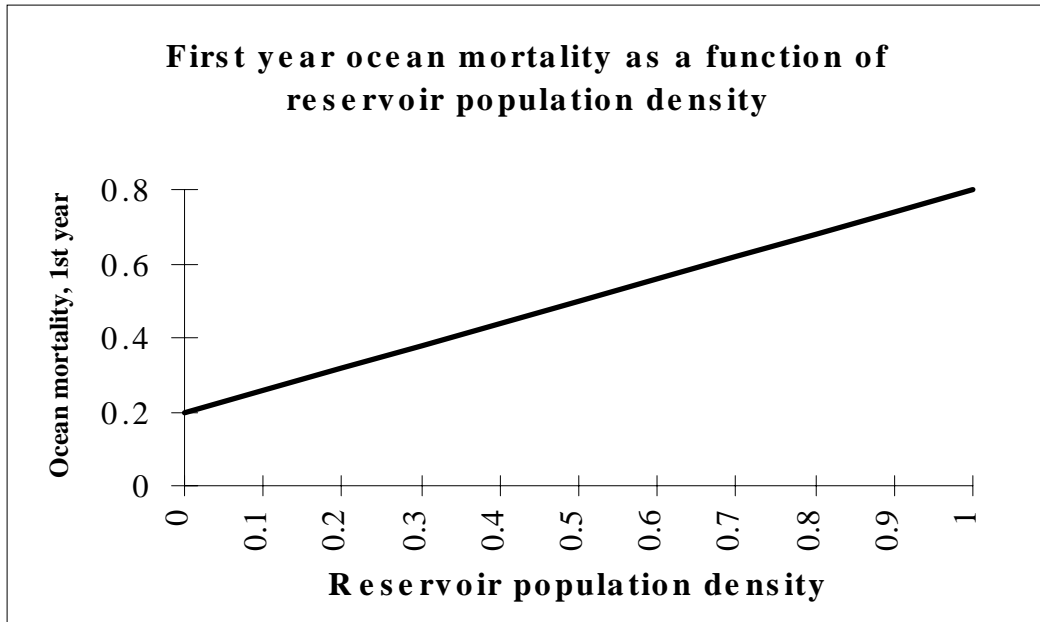


Figure 6. Example of first-year ocean mortality adjusted for sockeye population density, showing mortalities only by way of example. Actual mortalities that define the mortality function are defined by the user.

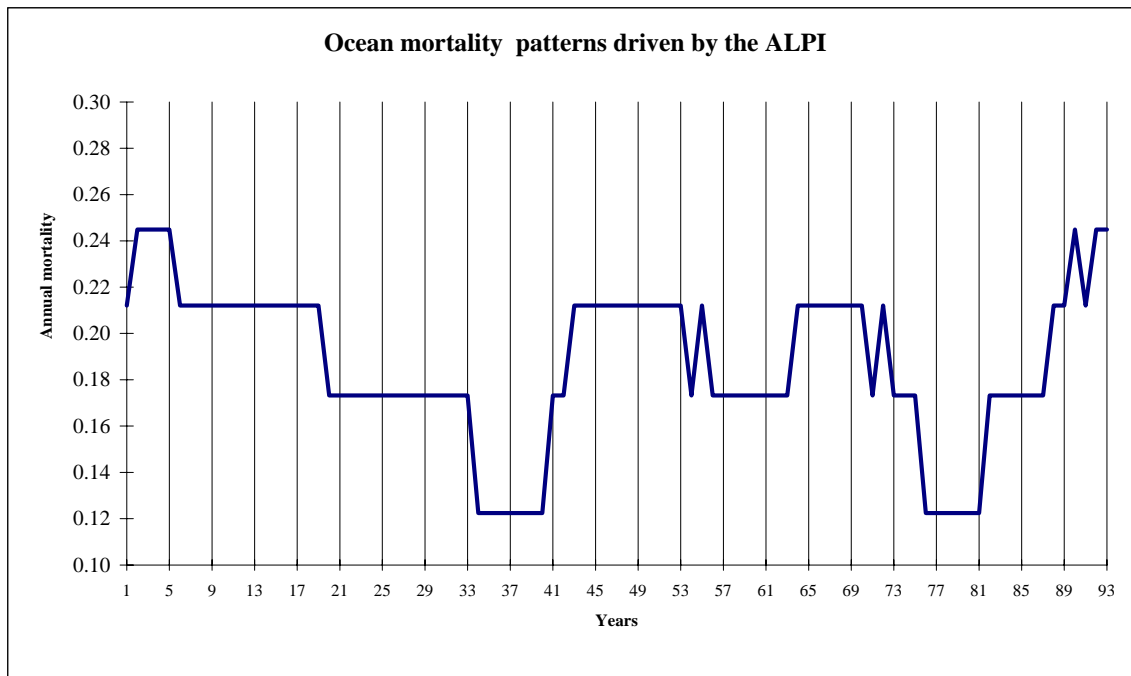


Figure 7. Ocean mortality as a function of the Aleutian Low Pressure Index. Vertical axis values are defined by the user.

Upstream Migration Stage

Each year, the group of surviving chinook or sockeye from each of the year classes that has reached maturity starts back up the Columbia. Mortalities for upstream passage are defined as: mortality while the fish are in the Columbia (primarily due to fishing and the two dams); mortality while the fish are in the Deschutes primarily due to fishing, mortality due to moving the fish past the three dams on the Deschutes; and mortality due to disease in the Deschutes. These mortalities are represented in the simulation by sampling from binomial distributions.

Fishing mortality can be modeled two ways: either as an ordinary, constant binomial parameter representing average annual mortality due to fishing, or as a constant escapement target. If the user selects the escapement target option, *PasRAS* will attempt to achieve that target by comparing the current spawner population with the escapement target, and assuming a level of error defined by the user. Fishing mortalities that in reality might happen in the ocean as well as in the Columbia or Deschutes are combined and implemented one time at the returning adults stage (just as the adults enter the Columbia River).

VALIDATION

PasRAS is not intended to be a predictive model, but rather a tool for exploring the relative impacts of various parameters. The approach taken to validation follows the recommendations of Jay Forrester and Peter Senge (Forrester and Senge 1980), two of the leading authorities on dynamic systems modeling.

Forrester and Senge define validation as “the process of establishing confidence in the soundness and usefulness of a model” (Forrester and Senge 1980, p. 414). Their validation process begins with the model builder developing confidence that the model is structurally sound and produces modes of behavior consistent with what is known about reality. The next step is to bring in experts not involved in building the original model, in order to test what Forrester and Senge call “transferable confidence” (Forrester and Senge 1980). They argue that “the ultimate objective of validation in system dynamics is transferred confidence in a model’s soundness and usefulness as a policy tool... We believe confidence is the proper criterion because there can be no proof of the absolute correctness with which a model represents reality. There is no method for proving a model to be correct... [thus] validity as meaning confidence in a model’s usefulness is inherently a relative concept” (Forrester and Senge 1980, p. 415).

It is interesting that for validation of dynamic models, Forrester and Senge do not rely on common statistical tests used, for example, to compare regression models to field data. The textbook approach to validation of models when the underlying mechanisms are well-understood is to compare model predictions to actual independent data. This can be a reassuring approach when such data are available; the problem is that when that kind of data is available, there is rarely a need for a dynamic simulation model. The kinds of problems that merit the effort required to develop and validate a model such as *PasRAS* are often the kinds of problems that do not have readily available data-sets that are complete enough to be useful in that way.

The implications of these limitations are that for *PasRAS*, as for any model, validation requires defining the contexts in which it would be valid to use the model. *PasRAS* was intended to be valid as a “thinking” tool, which could be used to examine relative impacts of risks at different life cycle stages, for a set of simple assumptions accepted as generally valid among experts on sockeye and chinook salmon. In order to develop what Forrester and Senge call “transferable confidence,” they recommend a variety of tests useful for examining model structure, model behavior, and policy implications. Working from Forrester and Senge’s recommendations for evaluating model structure, behavior, and implications, a list of eight questions was developed to summarize overall validity:

1. What characteristics of salmon life cycles have been left out or simplified, and what effects might those omissions or simplifications have on model output?
2. Is the structure of *PasRAS* adequate to serve the purposes for which it will be used?
3. Are the parameters and ranges used valid?
4. Does *PasRAS* produce long-term steady-state sustainability when it should, does it produce declines when it should, and does it produce extinctions when it should?
5. What are the anomalous or surprising behaviors that have been observed, and have they all been explained and/or corrected?
6. How do *PasRAS* structure and behavior compare to similar models? Does *PasRAS* address limitations in those models, and if so, how? Are limitations in *PasRAS* revealed by comparison to those models, and if so, how?
7. What parameters are *PasRAS* results most sensitive to? How do these sensitivities correspond to what is known about the real world?
8. How does *PasRAS* respond to simulated conditions or policy changes? In particular, (1) how does *PasRAS* respond to changes in marine and freshwater environments; (2) how does

PasRAS respond to changes in spawning and rearing habitat quality; (3) what kind of collection efficacies might be required in order to compensate for ocean mortality and less-than-optimal spawning and rearing habitat quality? Do these responses seem reasonable?

A literature review was conducted to establish ranges for parameter values. Literature that was used for validation scenarios is cited in Appendix B. Literature was also reviewed by Dr. Schreck (chinook version) and Dr. Mundy (chinook and sockeye species models; see Mundy 1998).

PasRAS was validated stepwise, starting with a baseline set of parameters defined by Dr. Mundy (sockeye) and Dr. Schreck (chinook). Next, these baseline parameters were modified in order to simulate several existing runs: Lower Granite on the Snake River, and Sherars Falls on the Deschutes River for chinook; the Kenai (Alaska), and Lake Osoyoos (above the Okanogan River in Washington) for sockeye. Finally, to examine the possible impacts of management decisions, specific scenarios were developed and evaluated. Summaries of the settings used in the run reconstruction and management scenarios are given in Appendix B.

Chinook Validation

The baseline scenario used for chinook was provided by Dr. Schreck, and was based on discussion with the PATH² team's studies of Snake River spring chinook. The chinook baseline scenario was selected because it was regionally applicable and also had the most widely-accepted parameter settings readily available.

BASELINE CHINOOK SCENARIO: LOWER GRANITE RUN RECONSTRUCTION

The baseline scenario parameter settings were calibrated using Lower Granite Dam (Snake River) escapements for 1975–1995 (Washington Department of Fish and Wildlife and ODFW 1996). Results were compared to simulated Snake River runs. Parameter settings are listed in Appendix B. Results are shown in Figure 8.

The average escapement for 1975–1995 was 18,339 (s.d.=11,542) for actual counts, and 16,456 (s.d.=2,640) for *PasRAS*. Actual and simulated Lower Granite average escapements were not found to be significantly different, even at the high α level of 0.2, using a t-test comparing means, assuming unequal variances. The mean *PasRAS* plot appears to be a good fit through

² PATH -- Plan for Analyzing and Testing Hypotheses -- is a group of federal, state and tribal scientists charged with gathering the scientific data that will assist policymakers in making key decisions in 1999 on the long-term configuration of the Columbia/Snake federal hydropower system.

actual Snake River escapements on the graph, though the correlation between the mean *PasRAS* plot and the actual run counts is only 0.5. The low correlation is partly because it represents a comparison of only 21 data points of a mean plot, with 21 points of a relatively stochastic empirical data set. Variability in the run counts is partly due to variability in harvest levels, as well as to the large Snake River runs in 1977–1978. These returns were from the last brood years before the last dam was built on the Snake River. This threshold event was not modeled in *PasRAS*.

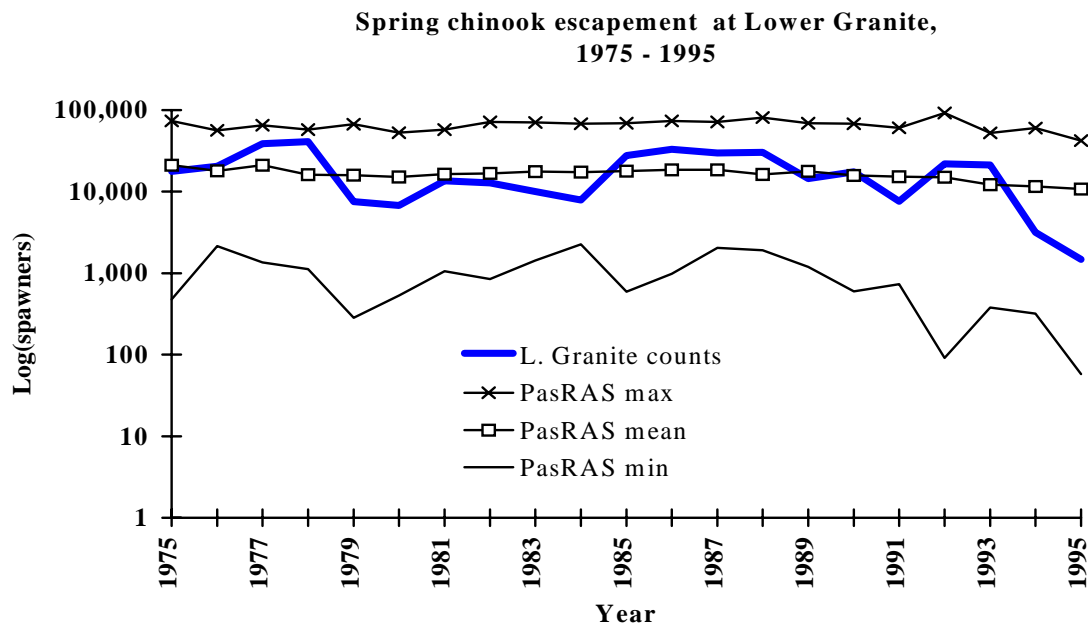


Figure 8. Spring chinook escapements at Lower Granite dam (Snake River), 1975–1995: actual and simulated.

DESCHUTES RIVER RUN RECONSTRUCTION

Deschutes River runs were reconstructed by modifying the Lower Granite scenario to account for several differences, including the fact that Deschutes River spring chinook counted above Sherars Falls have only two mainstem dams to negotiate, rather than the eight dams the Lower Granite fish negotiate. In addition, fecundity and age structuring were changed to reflect available data provided by Colleen Fagan, CTWS (Fagan 1998). Parameter settings are listed in Appendix B. Results were compared to run counts 1977–1998 above Sherars Falls, also provided by Colleen Fagan (Figure 9).

The average escapement for 1977–1998 was 1,727 (s.d.=714.5) for actual counts, and 2,217 (s.d.=560) for *PasRAS*. Actual and simulated Sherars Falls average escapements were found to be significantly different at the $\alpha = 0.1$ level, using a t-test comparing means, assuming unequal variances; *PasRAS* was on the high side of actual counts. Aside from being a little optimistic, the mean *PasRAS* plot appears to be a good fit through actual Deschutes River escapements on the graph, though the correlation between the two was only 0.5. As with the Lower Granite run reconstruction, the low correlation is partly because it represents a comparison of only 21 data points of a mean plot, with 21 points of a relatively stochastic empirical data set. Another source of variability in the actual run counts that was not present in the simulation is the fact that actual harvests varied from 0 to 81%, whereas *PasRAS* simply used the average harvest rate of 30%.

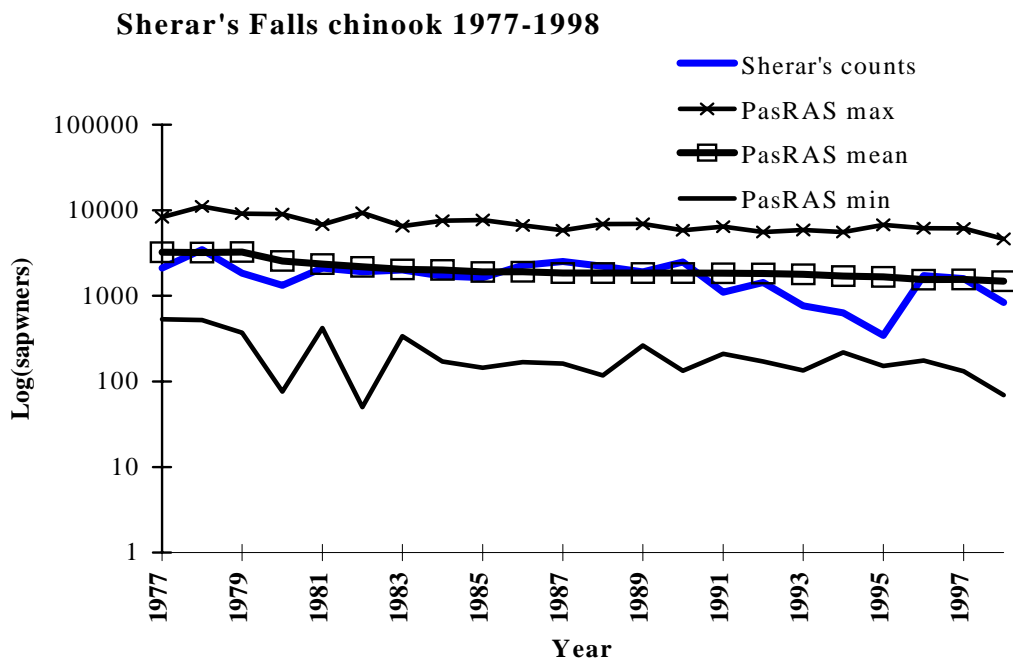


Figure 9. Spring chinook escapement above Sherars Falls 1977–1998: actual and simulated.

Because the “*PasRAS* mean” plots in Figure 8 and Figure 9 show the mean spawner populations from 500 replications, annual fluctuations due to model stochasticity are hidden. Figure 10 shows the same mean plot as Figure 9, along with some examples of single replications. Note that single replications exhibit fluctuations that look more like actual run counts, because actual salmon runs are, in effect, single replications themselves.

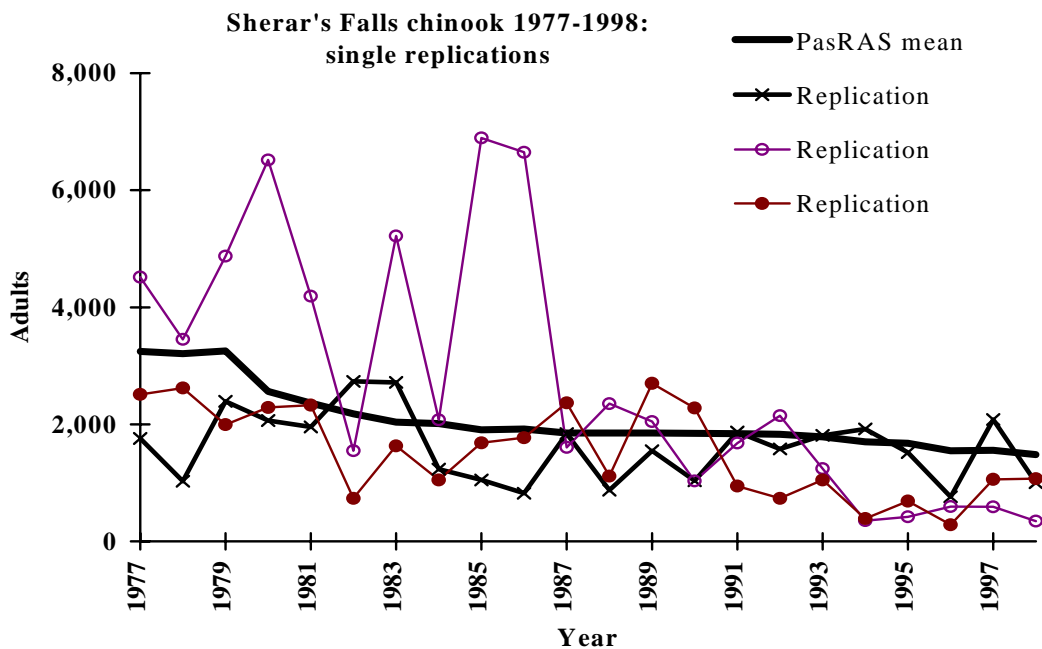


Figure 10. Sherars Falls chinook simulations, showing *PasRAS* mean over 500 replications and three single replications.

Sockeye Validation

BASELINE SCENARIO: KENAI RUN RECONSTRUCTION

A baseline set of parameters for sockeye was defined by Dr. Mundy, based on Foerster’s studies of Canadian sockeye (Mundy 1998, Foerster 1968). This set was assumed to represent a vigorous sockeye population living in the environment to which they had adapted over thousands of years, with excellent spawning and rearing habitat, and no dams. These parameters were used to define a baseline scenario that could be compared to actual Kenai spawner counts 1969–1990 [Alaska Department of Fish and Game (ADFG) 1998]. The model was calibrated by varying ocean parameters that were not known (e.g., annual ocean mortality by age and size of smolts, and starting point on the ALPI function), while maintaining an overall average of about 0.9 net weighted ocean mortality (Mundy 1998). *PasRAS* was run for 500 replications of 100 years each. The results, shown in Figure 11, are presented in log-transformed values because the scale is so large. The highest *PasRAS* value in 1987 over 500 replications was over 50,000,000 fish, for example. Untransformed graphs do not show this extent of variation well.

Kenai River sockeye runs, 1969-1990

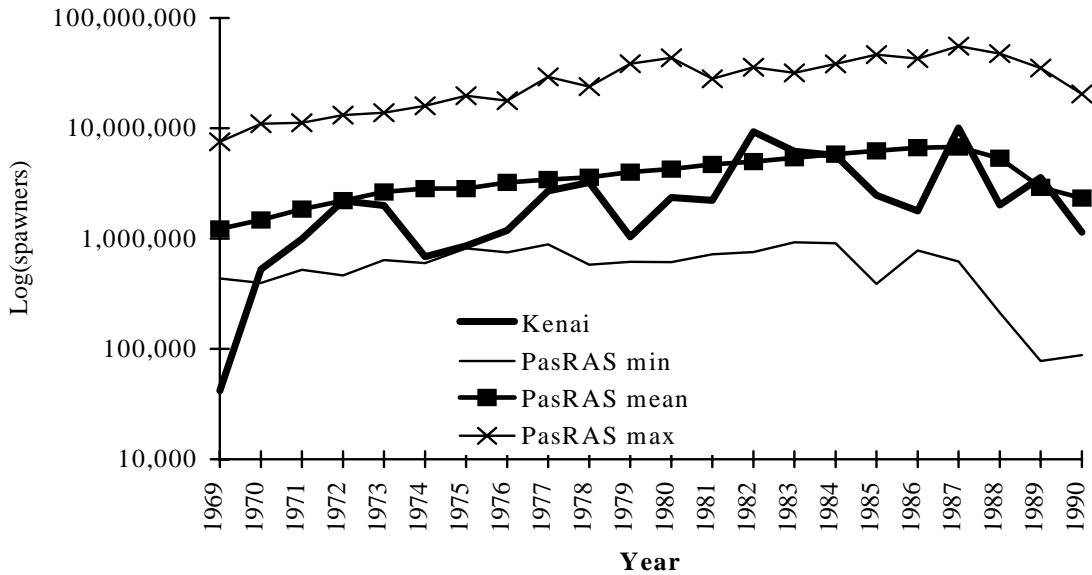


Figure 11. Kenai sockeye escapement, 1969–1990: actual and simulated. Graph shows log-transformed sockeye returns to the Kenai Peninsula, and the mean and range of 500 *PasRAS* replications.

The Kenai run reconstruction produced satisfactory results. The average escapement predicted by *PasRAS* was not statistically different from that predicted by ADFG data at the $\alpha = 0.1$ level (3,408,299 (s.d. = 5,104,941) for *PasRAS*, versus 2,826,966 (s.d.=2,681,720) for ADFG), and the general trends were reproduced fairly well. A linear regression of the predicted versus observed data points produced an R^2 of 0.72. This was not a bad result, considering it was comparing only 21 data points of a mean plot with 21 points of a relatively stochastic empirical data set.

COLUMBIA RIVER SOCKEYE RUN RECONSTRUCTION: OKANOGAN

A run-reconstruction scenario was also developed for the Okanogan Basin, a Columbia River subbasin (Figure 12). Okanogan fish spawn in the Okanogan River, upstream of Lake Osoyoos in British Columbia. Lake Osoyoos is eutrophic, shallow, warm, and highly productive (Fryer 1995); it may be more similar to Lake Billy Chinook than other Pacific Northwest sockeye rearing lakes such as Lake Washington or Lake Wenatchee (Mundy 1998–1999).

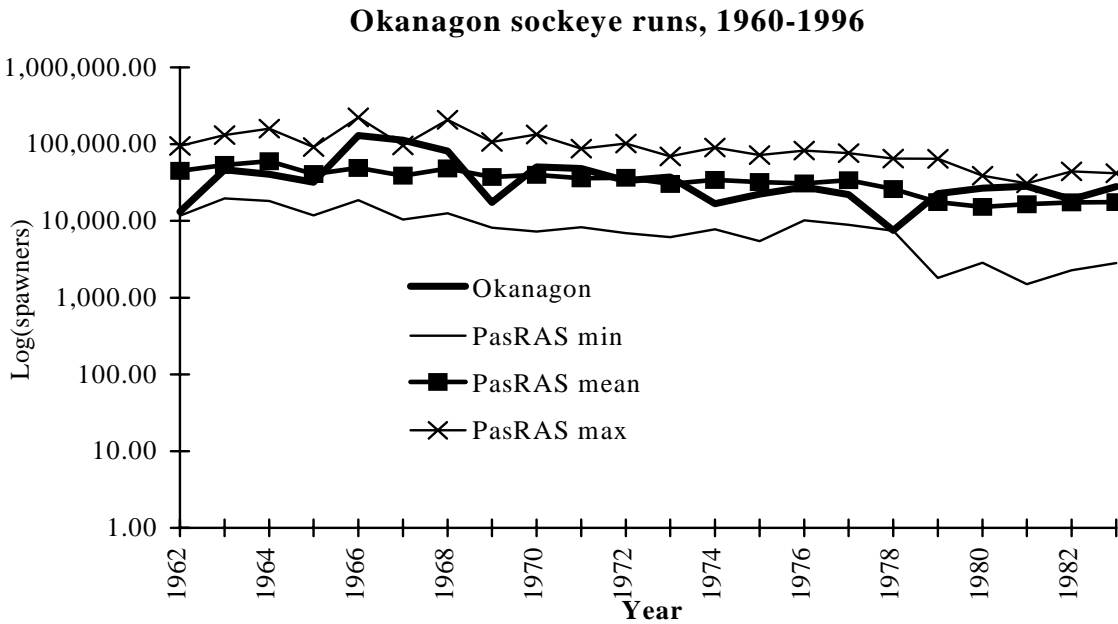


Figure 12. Okanagon sockeye escapement, 1960–1996: actual and simulated. Graph shows sockeye returns to the Okanagon Basin, and the mean, range, and confidence intervals of 1,000 *PasRAS* replications.

The Okanagon reconstruction used parameter values based on Jeff Fryer’s dissertation (Fryer 1995) and suggestions by Dr. Mundy. Extensive calibration and judgments were required for the density-dependent parameters for the Okanagon as they were for the Kenai. Actual run escapements were downloaded from Streamnet (<http://www.streamnet.org>); sources cited on the website were unpublished Washington Department of Fish and Wildlife reports.

Results were fairly comparable for the Okanagon reconstruction: actual and simulated average escapements over time were not found to be statistically different at the $\alpha \leq 0.2$ level: 34,851.89 (s.d. = 27,617) average escapement from the run counts, compared to 34,421 (s.d.= 12,540) simulated by *PasRAS*. A linear regression run between the actual and mean simulated escapements produced an R^2 of 0.86. This was not a bad correlation, considering the large variation of the actual data.

The minimum and maximum population size plots shown in both graphs indicate that *PasRAS* produced a wide range of variation over the 500 replications, as a good stochastic simulation should. The cyclic boom-and-bust patterns typical of sockeye populations are also well represented by *PasRAS* but are hidden within the mean plots. The longer-term patterns and

mean run sizes show in the mean plots, but the 1–10 year peaks and valleys are hidden. Untransformed plots of the Kenai run counts and five single replications show the boom-and-bust patterns of individual replication (Figure 13). These patterns can be produced by the fluctuation in first-year ocean mortalities assumed to be driven by the ALPI, although it is recognized that the ALPI may not have the same effect over all sockeye populations simulated.

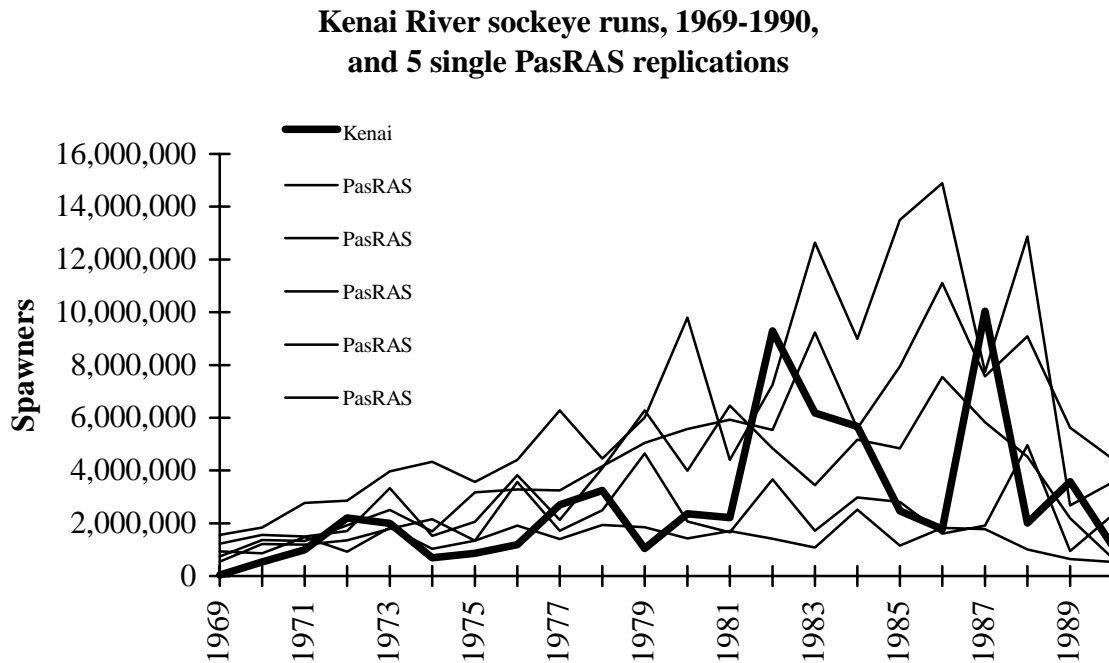


Figure 13. Untransformed actual Kenai sockeye run counts 1969–1990, and five single *PasRAS* replications.

In fact, it is not known whether the ALPI effect on *PasRAS* results could reasonably be assumed to be due to variations in ocean and climate, or whether it simply serves as a cyclic parameter that has an effect equivalent to some unknown function or process not otherwise included in *PasRAS* computer code. It is also possible to produce these kinds of boom-and-bust patterns via freshwater density dependence, even with “flat” (non-ALPI) ocean mortality. Freshwater density dependence serves as a negative feedback mechanism in that as modeled juvenile populations increase, so do their mortality rates; and delayed density dependence can serve as a negative feedback mechanism as well. There is considerable speculation about how real delayed density dependence might be (Myers et al. 1997, Myers et al. 1998, Welch and Mason 1990), but if it does exist, the assumption is that large juvenile populations over a year or two can lead to population crashes in subsequent years because of over-grazing of the zooplankton, and small juvenile populations can lead to larger populations in subsequent years because the zooplankton

have recovered. One of the sockeye versions of *PasRAS* does include delayed density dependence, but because it was so speculative, the sockeye workgroup decided it should not be used for these analyses.

It is important to emphasize, however, that it is actually easier to *not* reproduce Kenai or Okanogan runs with *PasRAS* than it is to reproduce them, since so few of the parameter settings were available from the literature. On the other hand, a comparison of survival, and cumulative survival, by stage, indicates that the assumptions required for reconstructing Kenai runs using *PasRAS* are not unreasonable (Table 1). For example, Table 1 shows that smolt-to-adult survival by *PasRAS* for the Kenai covered a range from 0.008 to 0.181 (midpoint 0.094), compared to 0.095 estimated by Foerster (1968) for the Kenai. Estimated sockeye egg-to-smolt survival in *PasRAS* is not so straightforward to calculate because of the feedback loops, but it varies between about 0.01 and 0.08 (midpoint 0.05); this is a little high, compared to Foerster's 0.01, but still reasonable. Foerster's Kenai sockeye exhibit approximately a 1.4 R/S ratio, whereas *PasRAS* simulations produced recruit/spawner ratios of from 0.17 to 22.69, not counting the feedback mechanisms.

As for the Okanogan, Fryer (1995) estimated smolt-to-adult average survival to be 0.011, whereas Mullan et al. (1992) estimated it to be 0.03; the *PasRAS* run reconstruction used between 0.008 and 0.026 (mean 0.017), well within the range of estimates provided by Fryer (1995) and Mullan et al (1992). Fryer estimated egg-to-smolt survival to be 0.073 to 0.25; Mullan et al. (1992) estimated average egg-to-smolt survival to be 0.12, with a range of 0.46–0.65. The range of egg-to-smolt values used by *PasRAS* for the Okanogan was 0.01 to 0.13 (midpoint 0.066).

Table 1. Cumulative survival: Comparison of *PasRAS* cumulative survival estimates with Foerster's recommendations. Total survival = adult returns/(effective spawners * fecundity). *R*= recruits. *P*=effective spawners (i.e., females). *f* = average fecundity. *S* = survival.

		Alaskan fecundity 3150 initially, then by age structure					Okanogon fecundity = 2900					
PasRAS stage	PasRAS "Kenai.stp"					Foerster		PasRAS "Okanagon"				
	Survival		Cumulative			Survival	Cumulative	Survival		Cumulative		
	Low	Hi	Low	Hi	Average			Low	Hi	Low	Hi	Average
egg - emergence	0.500	0.500	0.500	0.500	0.500	0.50000	0.50000	0.400	0.500	0.400	0.500	0.450
spring ("predation in tribes")	0.350	0.350	0.175	0.175	0.175	0.25000	0.12500	0.700	0.700	0.280	0.350	0.315
summer (constant predation, subject to constraint)	0.800	0.800	0.140	0.140	0.140	0.08000	0.01000	0.200	0.900	0.056	0.315	0.186
winter (density dependent)	0.100	0.600	0.014	0.084	0.049	1.00000	0.01000	0.100	0.400	0.006	0.126	0.066
collection	1.000	1.000	0.014	0.084	0.049	1.00000	0.01000	1.000	1.000	0.006	0.126	0.066
Deschutes	1.000	1.000	0.014	0.084	0.049	1.00000	0.01000	1.000	1.000	0.006	0.126	0.066
Columbia	0.950	0.950	0.013	0.080	0.047	1.00000	0.01000	0.350	0.350	0.002	0.044	0.023
Ocean (no fishing)	0.009	0.200	0.000	0.016	0.008	0.10000	0.00100	0.030	0.100	0.000	0.004	0.002
Columbia	0.950	0.950	0.000	0.015	0.008	0.95000	0.00095	0.750	0.750	0.000	0.003	0.002
Deschutes	1.000	1.000	0.000	0.015	0.008	1.00000	0.00095	1.000	1.000	0.000	0.003	0.002
Prespawn (in egg-emerg)	0.950	0.950	0.000	0.014	0.007	0.95000	0.00090	0.850	0.850	0.000	0.003	0.001
Total Survival/Generation	0.000	0.014			0.007	0.001	0.001	0.00004	0.003			0.001
Smolt-to-adult	0.008	0.181			0.094	0.095	0.095	0.008	0.026			0.017
Egg-to-smolt	0.014	0.084			0.049	0.010	0.010	0.01	0.13			0.066
Nominal R/P	0.17	22.69			11.43	1.42	1.42	0.05	4.08			2.07

ANALYSES

Chinook Analyses

POTENTIAL EFFECTS OF MANAGEMENT ACTIONS: SPRING CHINOOK

Jennifer Burke and Jeff Dambacher (Burke and Dambacher 1999; ODFW 1999) developed a habitat quality database called “*Habrata*” for the spawning and rearing habitat areas above Lake Billy Chinook. The database creates a text file, which the chinook species model can read to set up habitat-related parameters. The spring chinook simulations for Lake Billy Chinook used *Habrata*’s parameter settings as revised by Mike Riehle [USDA Forest Service (USFS)] for egg-to-emergence habitat quality, fry-to-smolt habitat quality, smolt capacity, and initial run sizes. The text file is listed in Appendix C.

It is important to note that the density-dependent parr survival curve used in the run reconstructions, referred to here as the “Baseline curve,” was provided by Dr. Carl Schreck (OSU). This survival curve was based on suggestions by Charlie Petrosky, for Snake River spring chinook. This curve is likely not completely accurate for Deschutes River spring chinook. It was hoped that density-dependent survival data from more local populations would be available. However, to directly use the data that were available for Warm Springs River spring chinook (Lindsay et al. 1989), it would have been necessary to lump the entire egg-to-smolt stage into one step. *PasRAS* allows the user to define two distinct stages for chinook:

1. Egg-to-emergence: non-density dependent mortalities, defined by egg-to-emergence habitat qualities;
2. Fry-to-smolt: density-dependent mortalities, defined by rearing habitat qualities.

Ordinarily, *PasRAS* determines mortality for the two stages according to the two habitat quality ratings defined by the user, but if the distinction between the two habitat qualities is lost, then it would assume the egg-to-emergence habitat quality to be the same as the fry-to-smolt habitat quality. Egg-to-emergence habitat quality is rated differently from rearing habitat quality in *Habrata* for most of the reaches: the correlation between the two is only 0.23, so lumping the habitat quality ratings together, which using the Lindsay et al. (1989) data would require, could produce significant error. The Baseline curve produced satisfactory Sherars Falls run reconstructions (Figure 9), whereas using the data from the Lindsay et al. (1989) study egg-to-smolt survival produced run sizes that were approximately 7,500 spawners larger.

The main reason for the difference is that Lindsay et al.’s (1989) study provided survival data on “deposition to migration” for Warm Springs chinook. He included not only spring migrants—

which were probably smolts—but also fall migrants—which had yet to over-winter, probably in the Deschutes River or perhaps the Columbia River. Lindsay et al. (1989) estimated that in 1978, fall migrant over-winter mortality was about 52%. Between 37% and 67% of the emigrants annually from the Warm Springs River were reported to be fall migrants. Consequently, the effect of assuming they all were smolts tends to significantly overestimate production. Lindsay (pers. comm. 1999) suggested some adjustments that might be tried to produce a fry-to-smolt survival curve that would be more consistent with expectations. Figure 14 shows the curve fits that result from regressing $\log(\text{survival})$ versus $\log(\text{relative seeding})$, to produce curves of the form *PasRAS* uses ($S_{parr} = K * F^{-r}$).

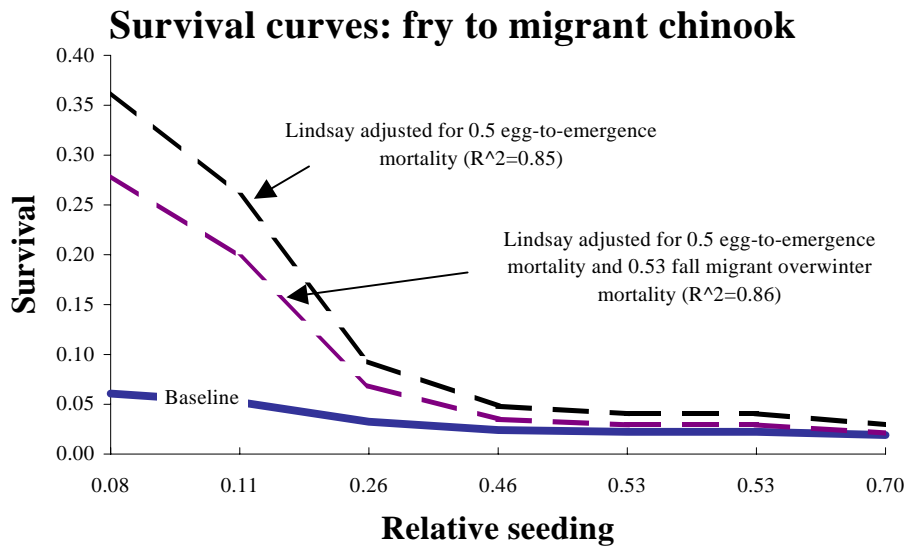


Figure 14. Fry-to-migrant chinook survival curve fits, comparing survival estimates resulting from the Baseline curve, and two adjustments to Lindsay et al.’s (1989) data: one that converted egg-to-migrant survival to fry-to-migrant by including a 0.5 egg-to-emergence mortality; and a second that adjusted further by including a 0.53 over-winter mortality.

Survivals using the adjusted Lindsay et al. (1989) data curve ranged from 50% (at a population density of 5% of full seeding) to 1.4% (at full seeding). Using the Baseline curve, survival would range from 7.8% (at a population density of 5% of full seeding) to 1.6% at full seeding.

Because the Baseline curve produced better results for the Sherars Falls run reconstruction, it was used for “Fair” rearing habitat quality in the management scenarios. For “Poor” habitat rearing quality, no density dependence was assumed (survival = 2%). For “Good” habitat rearing quality, the slope was increased from 0.53 to 0.60, increasing survival at full seeding from 1.6% to 2.5% and survival at 5% seeding to 15%. The main impacts of these changes are exhibited primarily below seeding levels of about 50% (Figure 14). Populations remained well below that

level in the simulations. Thus, assumptions about density dependence are very important for *PasRAS*, though it is far from clear how important they might be for natural populations, particularly at such low levels. Sensitivity analyses were also run (see Chinook Sensitivity Analysis) to illustrate the potential impacts of density dependence.

Chinook Scenario 0: No Management, 100% Collection Efficacy

Scenario 0 assumes:

- Runs were already established somehow, at an adult run size of 10% of the smolt capacity estimated by *Habrate* to be available in the Metolius, Crooked, and Deschutes River tributaries above Lake Billy Chinook [382,591 smolt capacity (per *Habrate*)]. The initial spawner escapement was reseeded if necessary to the initial value every year for the first three simulated years, with a C.V. = 0.15.
- The initial time period coincided with the initial ocean conditions shown on the ALPI graph, starting with relatively high mortality, with improving conditions through the next 50 years. This pattern was chosen because it is known that ocean survival is currently low, and some researchers believe that conditions are improving.
- Mortality due to stress or inadequate smoltification in the estuary was neglected.
- Juvenile collection efficacy was 100%.
- Harvest mortality was set at 30%, which is approximately the average level for Deschutes River chinook 1977–1998 (Fagan 1998).

For parameters that were not included in *Habrate*, the Sherars Falls run reconstruction settings were used (Appendix B). Results are shown in Figure 15.

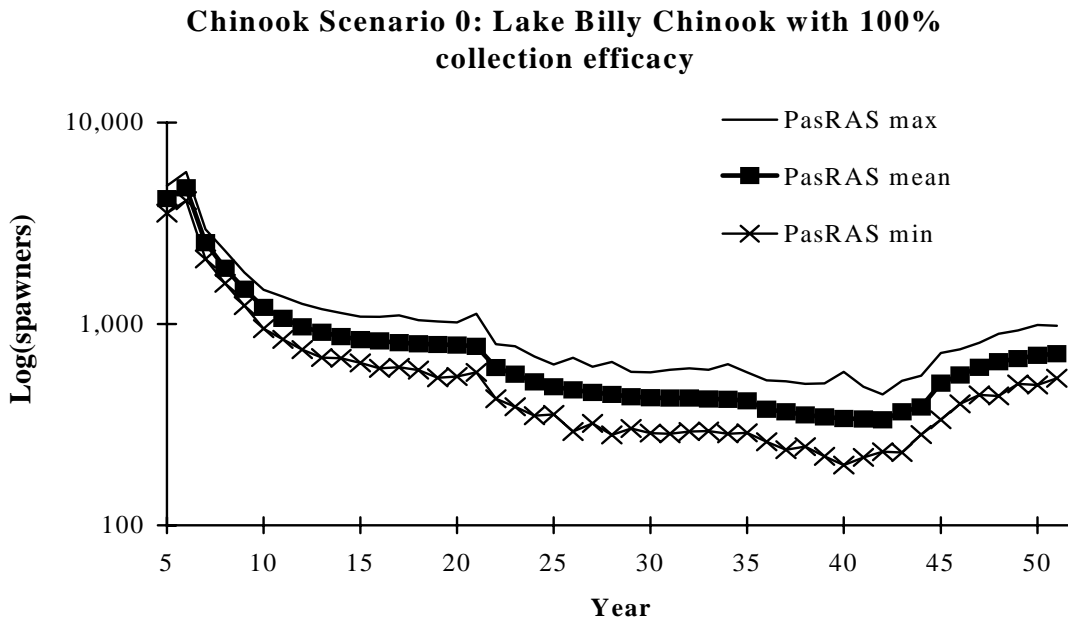


Figure 15. Chinook Scenario 0. Lake Billy Chinook using Habrate database settings, with 100% collection efficacy.

In Scenario 0, none of the 500 replications lasting 50 years each led to extinction. The average spawner population over the 50 years was 843 (s.d. = 878). The geometric mean³ recruits per spawner was 0.91, which is reflected in the overall declining trend. The simulation began with 85 reaches populated, and ended with an average of 6.81 reaches still populated. The reaches populated at the end of the simulation, along with the average ending spawner population, were (using the reach names assigned in *Habrate*):

- Crooked1: 27 spawners
- Crooked2: 254 spawners
- Crooked3: 152 spawners
- Crooked4: 61 spawners
- McKay7: 15 spawners
- Deschutes2: 86 spawners

³ The geometric mean is the nth root of the geometric sum of a series. For example, the geometric mean of 0.7, 0.8, 0.9 would be $(0.7 \cdot 0.8 \cdot 0.9)^{0.33} = 0.7958$.

Chinook Scenario 1: Effects of Reduced Collection Efficacy

The previous scenario assumed 100% juvenile collection efficacy, but it is not known how successful collection efforts might be. Estimates in the Columbia range from 20% to 60% juvenile collection and transport efficacy, though there is not much data on immediate or delayed juvenile mortality and National Marine Fisheries Service (NMFS) requires 95% collection efficacy (Tappel, pers. comm.). Scenario 1 used the same settings as Scenario 0, only with collection efficacy of 0.6. The results are shown in Figure 16.

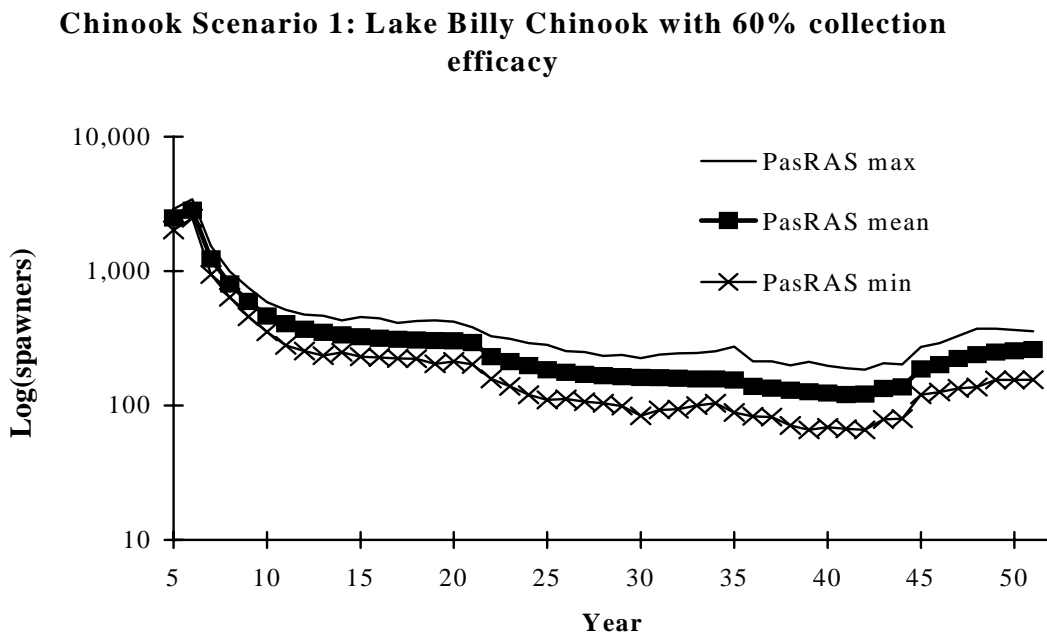


Figure 16. Chinook Scenario 1. Lake Billy Chinook using *Habrata* database settings with collection efficacy of 0.6.

In Scenario 1, none of the 500 replications lasting 50 years each led to extinction. The average spawner population over 50 years was 368 (s.d. = 528), compared to the average population with perfect collection efficacy (Scenario 0), of 843 (s.d. = 878) (significant at $p < 0.05$). The geometric mean recruits per spawner was 0.889, compared to Scenario 0's 0.91. The simulation began with 85 reaches populated, and ended with an average of 4.45 reaches still populated,

compared to Scenario 0's 6.81 reaches populated.⁴ The reaches populated at the end of the simulation, along with the average ending spawner population, were (using the reach names assigned in *Habrata*):

Crooked2: 122 spawners

Crooked3: 52 spawners

Crooked4: 16 spawners

Deschutes2: 62 spawners

CanyonCreek1: 4 spawners

Reducing collection efficacy by 40% (from 100% to 60%, a number that may be optimistic on the Columbia), reduced the average spawner population by 56% and also reduced the probability of population persistence; to survive over the long-term, the geometric mean R/S needs to be over 0.9, preferably at least 1.0. With either perfect or reduced efficacy, the geometric mean R/S was hardly sustainable, at about 0.89. The last several years of the simulation enjoy relatively good ocean conditions because the simulation runs during an improving series of ALPI periods.¹ Simulations were started at the point where ocean conditions were bad, but improving, which is what researchers believe may currently be the case for Columbia River fish; thus, with only four or five reaches still populated, by an average of about 256 total fish, it is possible that this simulated population might be a candidate for listing under the Endangered Species Act.

Chinook Scenario 2: Effects of Initialization.

The previous scenarios assumed chinook runs had already been established at the level defined as full seeding by *Habrata*. To seed the simulation, it was assumed that initial spawner populations would correspond to smolt-to-adult survival = 0.1 (38,259 adult fish) at full seeding—no doubt an optimistic assumption. In reality, it may take many years to establish successful fish passage at the PRB project, and the fish available for establishing the runs may come from Deschutes River hatcheries. If so, it might mean that initial populations will have to be small because Deschutes River hatcheries have seen escapements since 1993 of less than 2000 adults (Fagan 1998).

Perhaps a more realistic scenario would be to start the simulation by assuming that all spring chinook that return to the Pelton trap could be used to initiate fish passage at PRB. To do this,

⁴ Statistically comparing the geometric means and number of reaches populated at the end of the simulations requires recording more output than was done during the scenario analyses.

the simulation was stochastically re-seeded every year for 3 years by sampling from a normal distribution where the mean and s.d. would be the mean (818) and s.d (641) of the Pelton returns since 1977 (Fagan 1998). The Riehle version of *Habrate* was used for parameter settings, except that instead of the default seeding levels of 38,259 fish, initial seeding was assumed to be the 818 fish from the average Pelton trap returns. In order to link them to the habitat ratings and smolt capacity estimates by reach, their offspring were assumed to distribute themselves stochastically through the 92 reaches proportionate to the reaches' smolt capacity.

Scenario 2 otherwise assumed the same parameter settings as Scenario 1, using for collection efficacy the 60% estimate (Tappel 1998). The results are shown in Figure 17.

In Scenario 2, none of the 500 replications lasting 50 years each led to extinction. The average spawner population over the 50 years was 220 fish (s.d. = 72). The geometric mean recruits per spawner was 0.99. It is interesting that the R/S actually increases when the initial populations are smaller. This is the kind of impact that the density dependence assumption has: because fry-to-smolt survival increases at low population densities for fair and good rearing habitat, the net effect is to increase resilience, as reflected by the R/S ratio. Whether this effect reflects reality is another question altogether.

220 fish is significantly ($p < 0.05$) less than the average population of 368 from Scenario 1. However, the difference disappears by about the 15th year, and the 50th year populations are almost exactly the same, regardless of starting population size. The implication is that initiating the simulation with larger populations has no effect on how the simulation ends; this is probably because the initial 85 reaches decreases within a few years to about 4.3 reaches regardless of assumption, and because the steepness of the parr survival curve compensates for low population densities.

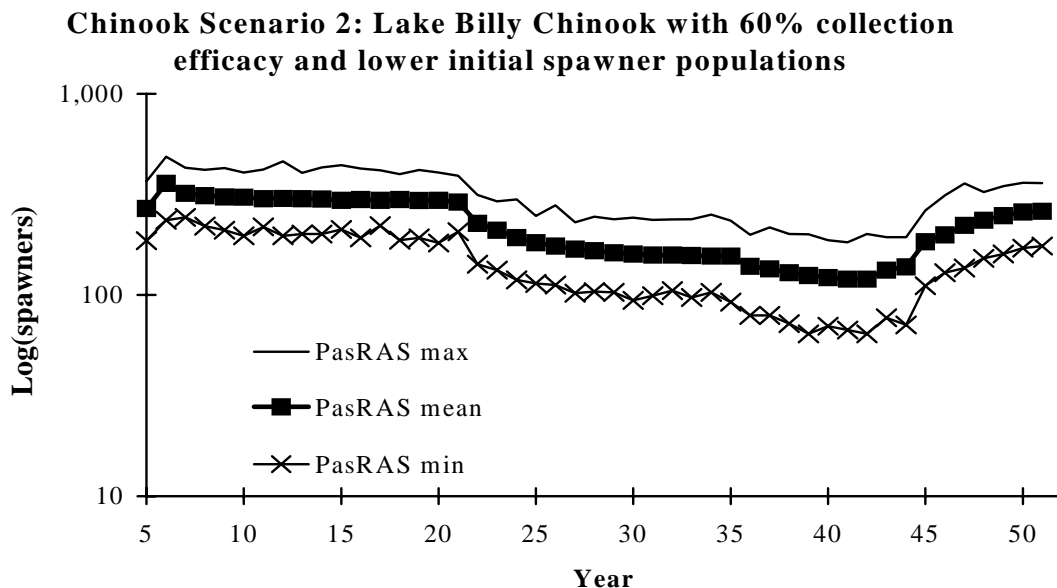


Figure 17. Chinook Scenario 2: Establishing a new run by supplementing the first 3 years with historical Pelton trap return estimates.

The reaches populated at the end of the simulation, along with the average ending spawner population, were (using the reach names assigned in *Habrata*):

Crooked2: 141 spawners

Crooked3: 73 spawners

Crooked4: 16 spawners

Deschutes2: 60 spawners

Chinook Scenario 3: Effects of Smolt Capacity

The density-dependent chinook parr survival function depends on the definitions of smolt capacity in *Habrata*, which reflect rearing habitat quality. Concerns had been expressed by the FTS that these estimates might be high (the total smolt capacity for the tributaries to Lake Billy Chinook added up to 382,591 smolts). Scenario 3 used the same settings as Scenario 0 (i.e., perfect collection efficacy), except with smolt capacity estimates from *Habrata* reduced by 20%. Since the simulations are initialized by assuming initial returning spawner populations of

0.1*smolt capacity, this also reduced the initial population sizes for the first three years (recall that *PasRAS* supplements returns for the first three or more years in order to allow age structuring to become established).

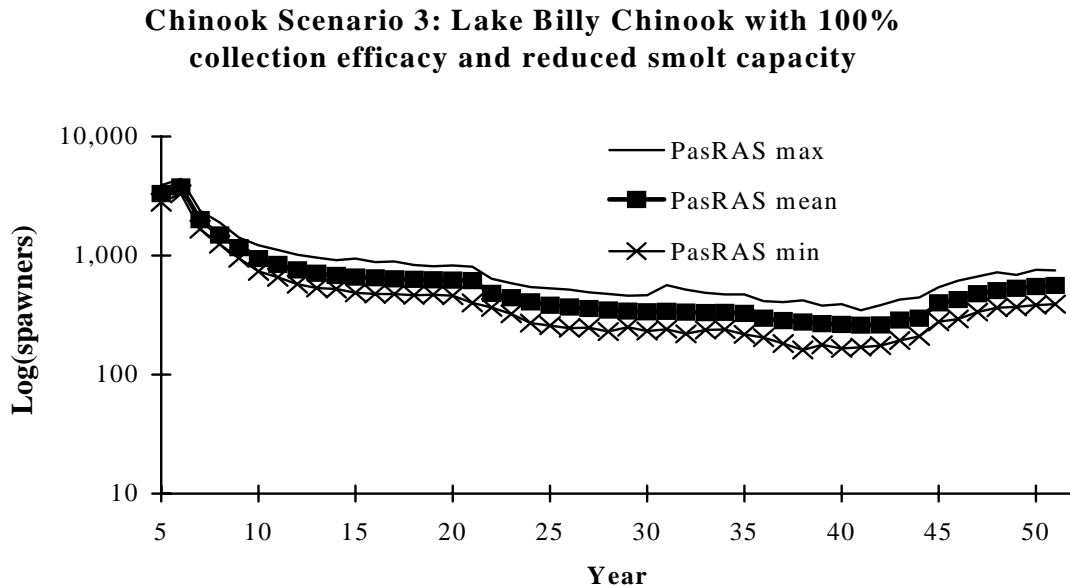


Figure 18. Chinook Scenario 3: Lake Billy Chinook using *Habrata* database settings with collection efficacy of 1.0 but reducing smolt capacity by 20%.

The results are shown in Figure 18. In Scenario 3, none of the 500 replications lasting 50 years each led to extinction. The average spawner population over 50 years was 664 fish (s.d. = 697). The geometric mean recruits per spawner was 0.912. The result suggests a linear impact: decreasing smolt capacity by 20% decreased the average population by 21%. It had no effect on the geometric mean recruits per spawner and only reduced the average number of reaches populated by 0.74 (from 6.81 to 6.07, probably not statistically significant although data were not collected to allow a t-test). The reaches populated at the end of the simulation, along with the average ending spawner population, were (using the reach names assigned in *Habrata*):

Crooked1: 22 spawners

Crooked2: 231 spawners

Crooked3: 96 spawners

Crooked4: 50 spawners

McKay7: 10 spawners

Deschutes2: 130 spawners

CanyonCreek1: 40 spawners

The reason decreasing smolt capacity affected neither extinction probability nor population robustness is probably because juvenile survival varies so strongly with population density, and anytime the population approaches extinction, survival goes up enough to generally compensate.

Chinook Scenario 4: Results Using Adjusted Lindsay et al. (1989) Data for Fry-to-Smolt Survival

Although more extensive sensitivity analyses were conducted to illustrate the impacts of the density-dependent parr survival curve, a single simulation was also run to provide a direct comparison of results using the Baseline curve to results using the curve derived using adjusted Lindsay et al. (1989) data (discussed above). Recall that the two survival curves are (Figure 14):

$$Sparr_{\text{Baseline}} = 0.016 * F^{-0.53}$$

$$Sparr_{\text{Lindsay}} = 0.014 * F^{-1.193}$$

Scenario 1 was repeated, using the Lindsay et al. (1989) curve. The results are shown in Figure 19.

Over the simulated 50 years, the average spawner population size in Scenario 4 was 1,114 (s.d. = 384), and there were no extinctions. An average of 9.45 of the initial 85 reaches were still populated, and the geometric mean R/S was 0.966. The reaches populated at the end of the simulation, along with the average ending spawner population, were (using the reach names assigned in *Habrate*):

Crooked1: 42 spawners

Crooked2: 132 spawners

Crooked3: 302 spawners

Crooked4: 175 spawners

McKay7: 25 spawners

Deschutes2: 346 spawners

CanyonCreek1: 43 spawners

CanyonCreek2: 15 spawners

LinkCr1: 13 spawners

These results were significantly more optimistic than those achieved using the Baseline curve: the final population almost tripled, the number of reaches populated in the 50th year doubled, and the geometric mean R/S increased from a marginally sustainable 0.89 to a probably sustainable 0.96. Using the Lindsay et al. (1989) data did not produce Sherars Falls run reconstructions that were as good as reconstructions using the Baseline curve.

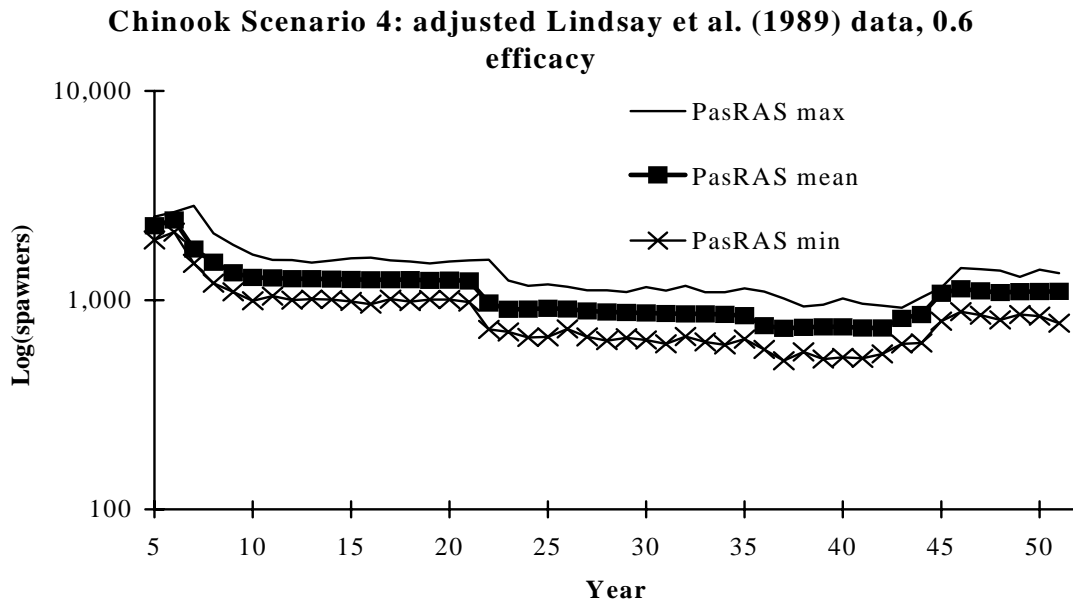


Figure 19. Chinook Scenario 4: results using adjusted Lindsay et al. (1989) data for parr survival curve (compare to Scenario 1, Figure 16). Scenario plots start at year 5 because it takes 5 years to establish age structuring.

Summary of Chinook Scenario Results

A comparison of the results from the above scenarios is shown in Table 2.

Table 2. Summary of Chinook scenario results.

Scenario	Average spawner population over 50 years (s.d.)	Geometric mean R/S (500 replications)	Average reaches still populated, year 50
0. 100% collection efficacy, start at full seeding, supplement as needed for 3 years	843 (878)	0.913	6.81
1. 60% collection efficacy, start at full seeding, supplement as needed for 3 years	368 (528)	0.889	4.45
2. 60% efficacy, start with Pelton Trap returns only (2% of full seeding as defined in Habrate) and supplement as needed for 10 years	220 (72)	0.990	4.37
3. 100% efficacy, start at full seeding, supplement as needed for 3 years, but reduce smolt capacity by 20%	664 (697)	0.912	6.07
4. 60% efficacy, start at full seeding, supplement as needed for 3 years, and use adjusted Lindsay et al. (1989) data for fry-to-parr survival	1,114 (384)	0.966	9.45

The differences among the spawner populations summarized in Table 2 are all statistically significant at the 0.05 level.

Although there were no extinctions in any of the chinook scenarios, it is worth noting that the geometric mean R/S ratios were all below the nominal replacement value 1.0. The scenario that is probably the most relevant to chinook reintroduction, Scenario 2, had the best R/S (0.99). Average spawner population sizes tended to be small (843 for 100% collection efficacy but in the ~200–400 range for the two scenarios that are most relevant). Also of concern is that although the simulations all began with 85 reaches populated by the *Habrate* database settings, only 4–7 on average were still populated after 50 years. The more optimistic exception to these observations is Scenario 4, which was a repeat of Scenario 3 but with the more optimistic density-dependent parr survival curve derived from Lindsay et al.’s (1989) data.

Table 3 lists a summary of 50th-year average populations by reach. The most robust reaches were Crooked River reaches 2, 3, and 4, and Deschutes River reach 2, which always ended up

with spawning populations. Crooked River reach 1, McKay reach 7, and Canyon Creek 1 were more marginal: they maintained populations when conditions were relatively better (e.g., high collection efficacy and high density dependence) but otherwise had their populations extirpated. Canyon Creek 2 and Link Creek 1 only maintained populations when density-dependent survival was increased by using a relationship based on Lindsay et al. (1989).

Table 3. Summary of chinook population results for reaches still populated after 50 years.

Reach name	Scenario 0 average spawners in 50 th year	Scenario 1 average spawners in 50 th year	Scenario 2 average spawners in 50 th year	Scenario 3 average spawners in 50 th year	Scenario 4 average spawners in 50 th year
Crooked1	27	0	0	22	42
Crooked2	254	122	141	231	132
Crooked3	152	52	73	96	302
Crooked4	61	16	16	50	175
McKay7	15	0	0	10	25
Deschutes2	86	62	60	130	346
CanyonCr1	55	4	0	40	43
CanyonCr2	0	0	0	0	15
LinkCr1	0	0	0	0	13

It may seem surprising, because the stream seems to be of such generally high quality, that none of the Metolius reaches ended up with chinook populations. Mike Riehle, who adapted *Habrata* for these simulations, explained that the reason for this result is that under the rules of the *Habrata* habitat rating system [developed by Burke and Dambacher (1999)], chinook rearing only occurs in pool habitat. The USFS protocol only considered pools if they were as wide as they were long. Short pools are common in broad, spring-fed channels of the Metolius system, but these pool habitats or side pool habitats in the Metolius were not included in the total percentage of pools, and may have resulted in a lower rating of habitat quality for the Metolius River. Also, older surveys conducted by the USFS and current ODFW protocols inventory pools and glides separately. In of 4th and 5th order stream channels with good depth, these glides may provide some habitat for rearing chinook. Deep glide habitats were not included in the percent of pool habitat in *Habrata* and may have lowered the habitat quality rating for chinook in some reaches.

CHINOOK SENSITIVITY ANALYSIS

Both species models of the software provide the capability of performing multivariate sensitivity analyses, which can also be used to increase stochasticity for many of the parameters.

Stochasticity can be increased via two mechanisms: (1) for a parameter that is defined by a normal distribution, either the coefficient of variation or the standard deviation can be increased, and/or the mean can be expressed as a range; (2) for a parameter that is defined by a discrete probability, the nominal probability can be represented by a *range* of probabilities defined by minimum and maximum values.

Each time the simulation requires one of the parameters selected for sensitivity analysis, it first selects a value from the defined range either by essentially flipping a coin to choose the high or low end of the range, or by sampling from a uniform distribution defined by the range⁵. It then uses that new value instead of the previously-defined nominal probability. *PasRAS* next samples from a binomial distribution where \hat{p} = the new value that resulted from the coin flip, instead of the nominal probability. In this way, the probability \hat{p} that defines any of these binomial distributions will only rarely happen to be the nominal value. If the user picked the coin-flip option, then the probability used will instead be the minimum defined by the user about half the time, and the maximum defined by the user about half the time. If the user picked the uniform distribution option, then the probability used over all the replications will cover the entire range between the minimum and maximum defined by the user.

If the user instructs *PasRAS* to conduct sensitivity analysis, *PasRAS* records in an output file the actual values used for the selected parameters every time they are called for, as well as the spawner population sizes and recruits per spawner that result. This output file, named *sensitivity.txt*, is in CSV (comma-separated variable) format so that it can easily be imported into Microsoft Excel® or other analytical programs. There it can be statistically analyzed, using, for example, Analysis of Variance (ANOVA) in order to evaluate the relative impact of the uncertainty ranges on the outcomes (Fahrig 1991). Because variance is a good indicator of uncertainty, Analysis of Variance is a very useful tool for prioritizing data needs. Conducting multivariate sensitivity analyses in this way is much more efficient than one- or two-factor-at-a-time approaches, and it allows the user to evaluate the impacts of interactions among variables if desired.

⁵ If the user wishes to perform a full factorial experiment, the coin-flipping option should be chosen in order to enhance the contrast between the two levels. If the user wishes to assess the sensitivity of the results over a range of defined values, or simply to increase stochasticity in the simulation, then the uniform distribution option should be chosen.

A convenient way of conducting Analysis of Variances for factorial experiments is to use Excel's regression tool. It is important to note, however, that in order to use this technique, the data has to be converted to the standard score form. For example, if Z is the standardized form for variable X , \bar{X} = the mean of X , and s = standard deviation of X then:

$$Z = \frac{\bar{X} - X}{s}$$

Once this conversion is made, as long as all the variables in the factorial experiment are orthogonal (not correlated with one another), then the analysis can be shown to be the same as performing Analysis of Variance (Ferguson and Takane 1989, p. 508). Regression coefficients calculated for the standardized data represent the correlations of the independent variables with the dependent variable, and the squared coefficients represent each variable's share of the total variance of the response variable.

Multivariate chinook sensitivity

Stochasticity due to ocean parameters was minimized by disabling the ALPI mortality option, disabling the option of adjusting return times according to ocean conditions, and setting all variances to zero. Otherwise, the settings used for the chinook sensitivity analysis were the same as for the Lake Billy Chinook simulations. The ratio of the high to low settings was held constant at 1.25 in order to help ensure that the relative impacts of the parameters would not be artificially biased towards those that had larger ranges. "Density dependence," which is the slope of the parr survival curve, is not a mortality per se as are the other parameters, but rather the parameter of a function that relates mortality to population density. The parameter ranges used for the chinook sensitivity analysis are shown in Table 4.

Table 4. Parameter ranges for multivariate sensitivity analyses for chinook salmon.

Parameter	Min	Max	Max/Min
Juveniles in tributaries	0.60	0.75	1.25
Density dependence	-0.49	-0.62	1.25
Collection efficacy	0.56	0.70	1.25
Spawner success	0.76	0.95	1.25
Estuary	0.18	0.22	1.25
Juv. Columbia losses	0.16	0.20	1.25
Adult transport Loss	0.112	0.140	1.25

The results are shown in Table 5. Coefficients that are shown in the table result from a regression that was conducted on data that had been transformed to the standard-score form as discussed above (Ferguson and Takane 1989). When regression is used in this way to conduct Analysis of Variance, it is necessary to show that the transformed parameters are orthogonal. This was done by calculating a correlation matrix, where the highest magnitude correlation was found to be 0.04.

Aside from the ocean parameters, the most critical (at $p < 0.05$) parameters for spring chinook were found to be (in order of effect): juvenile losses in the tributaries, the slope of the parr survival curve (“Density dependence”), juvenile collection and transport efficacy, prespawning mortality (“Spawner success”), mortality in the estuary, juvenile losses migrating through the Columbia and adult transport. Table 5 lists the relative ranking of the significant parameters indicating their relative impacts on the outcome. Virtually identical results are produced by analysis using Spawner population size as the response variable. Numerous sensitivity analyses had indicated that these seven freshwater parameters were the most significant; no other freshwater parameters were found to contribute significantly to outcome variance.

About 70% of the outcome variance is explained by the analysis ($R_{adj}^2 = 0.71$). The “contribution to variance” column in Table 5 indicates that over half is explained by juvenile mortality (0.45/0.71) in the tributaries. Most of the rest is explained by the slope of the density-dependent parr survival curve (0.107) and juvenile collection efficacy. Type II error was found to be less than 0.0005 (Montgomery 1985). This means that the probability of incorrectly failing to reject the null hypothesis (i.e., the probability of concluding that one of the parameters has negligible effect on recruits per spawner—and being wrong) is less than 0.05%.

Table 5. Analysis of Variance for multivariate sensitivity analysis for chinook salmon, with geometric mean R/S as response variable.

Geometric mean R/S	Coefficients	P-value	Contribution to variance
Juveniles in tributaries	-0.667	0.000	0.445
Density dependence	-0.327	0.000	0.107
Collection efficacy	0.275	0.000	0.076
Spawner success	0.217	0.000	0.047
Estuary	-0.054	0.002	0.003
Juv. Columbia losses	-0.054	0.002	0.003
Adult transport Loss	-0.047	0.006	0.002

Juvenile mortality in the tributaries, density-dependent survival of juveniles, and spawner success are indicators of tributary habitat quality. Estuary mortality is likely related to inadequate smoltification and/or stress due to juvenile collection and transportation timing and methods. Juvenile mortality in the Columbia is related to downstream passage through the two dams. Adult transport loss is due to transporting adults past the PRB project and any other mortality suffered by the adults on their return trip up the Deschutes, including fishing.

Thus, the most influential freshwater parameters in the chinook simulations were found to be:

1. Tributary habitat quality
2. Juvenile collection efficacy and quality (as it affects juvenile mortality, smoltification and stress)
3. Columbia River dam mortalities
4. Mortalities suffered by adults returning up the Deschutes

It is important not to forget that the sensitivity analyses were for *freshwater* parameters. Ocean parameters (mortality by age, age structuring, fecundity and length as functions of age) are responsible for most of the variance in any scenario simulated, and it is this stochasticity that leads to higher rates of extinction than would otherwise be indicated in non-stochastic analyses.

In addition to the multivariate sensitivity analysis, two single-parameter analyses were conducted. The FTS had originally been formed to evaluate the feasibility of fish passage and so one of the primary questions was: “how good might collection efficacy have to be?” There was thus particular interest in how the results of the model vary with juvenile collection efficacy. In addition, assumptions about density dependence proved to be critical in model performance, while there was unfortunately little data available in the literature to support parameter settings used. The intensity of impact of density dependence is determined by the slope of the parr survival curve, and thus a single-parameter sensitivity analysis was conducted for it.

Chinook Single Parameter Sensitivity 1: Effects of Collection Efficacy.

In order to get an idea of how *PasRAS* results vary with juvenile chinook collection efficacy, the sensitivity analysis “uniform” option was used. Efficacy was set to a uniform distribution that ranged between 0.4 and 0.9, rather than to a coin-flip choice between the “high” and “low” settings as was done for the factorial experiments described in the previous section. *PasRAS* was run with the Scenario 0 settings for 500 replications. The result is shown in Figure 20.

Figure 20 shows both the geometric mean R/S and the spawner populations after 100 years. The linear curve fit for the spawner population result was a better fit than for R/S ($R^2 = 0.78 > 0.14$; $p < 0.00005$ for both). The graph indicates that the population size in the 100th simulated year is sensitive to collection efficacy, all other things being equal: increasing collection efficacy from 40% to 90% (2.25-fold increase) produces about an 18-fold increase in average spawners (from 45 to 830) after 100 years. Collection efficacy had little effect on R/S. This is most likely because, in the simulation, ocean mortality has a much greater impact on the rate at which offspring survive to return and spawn. The low R^2 indicates that only 14% of the change in R/S from low to high collection efficacy is accounted for by the regression (i.e., collection efficacy). Visual inspection of the graph also makes clear that the five early datapoints where R/S went to 0 would significantly affect R^2 ; these occurred during the early part of the simulation, when ocean mortality was relatively high.

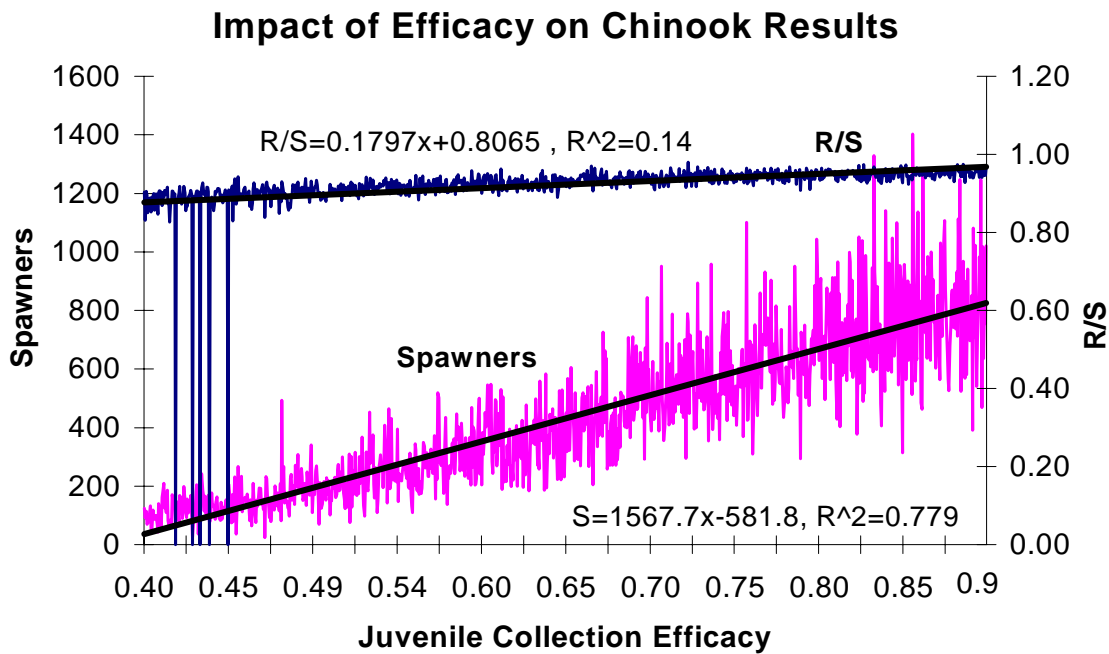


Figure 20. Impact of juvenile collection efficacy on chinook results.

Chinook single parameter sensitivity 2: Effects of density dependence.

If it is steep enough, the density-dependent survival function for parr can almost prevent extinction because parr survival increases exponentially as the population declines. A steep parr survival indicates that fry-to-smolt mortality is density dependent, and mortality at that stage will then tend to compensate for losses at any other stage of life in the model. Mortalities also

increase in the model at low population densities due to the genetic fitness mechanism, but this effect is small.

The parameters used for the parr survival function in the Snake River and Deschutes run reconstructions, and in the above scenarios, were the Baseline curve provided by Carl Schreck. As noted above, the Baseline curve produces significantly more conservative results than using a density-dependent egg-to-smolt survival curve derived from Warm Springs River data (Lindsay et al. 1989). To illustrate the effect of density-dependent parr survival (and hence of rearing habitat quality), the sensitivity analysis “uniform” option was used. The slope of the parr survival curve for “Good” rearing habitat was set to vary from -0.4 to -0.6 (the Baseline nominal value is -0.56). Minimum survival—survival at full seeding—was held constant at 0.016, so the only effect the variation would have would be on survival at seeding levels less than about 0.5. Collection efficacy was set to 100%, and all the reaches were rated as “Good” so that they would all have the same parr survival curve. The results are shown in Figure 21.

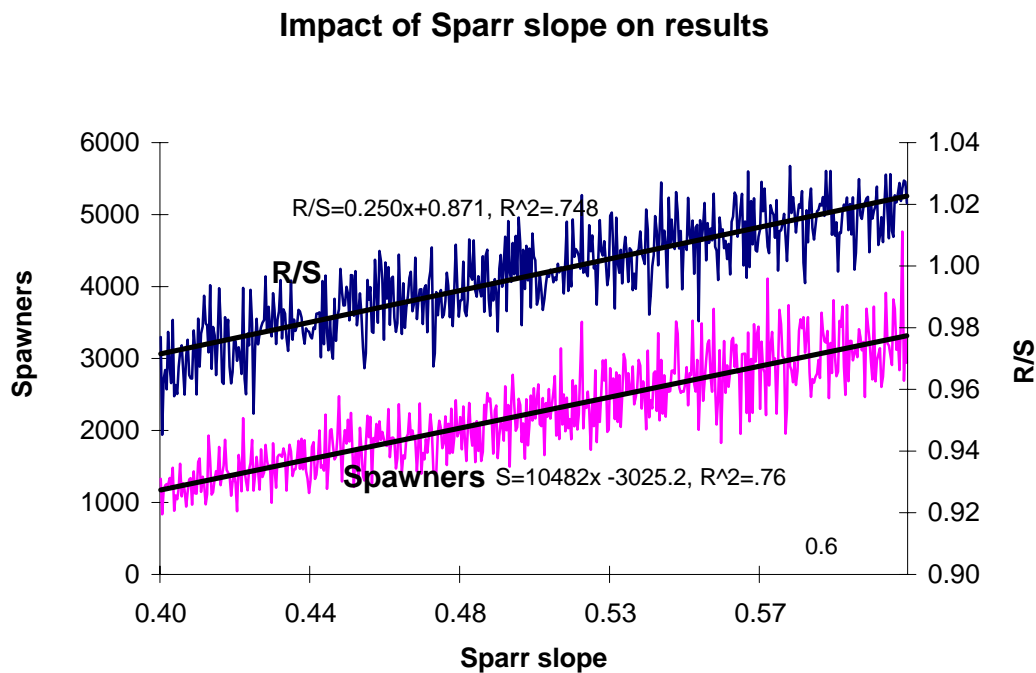


Figure 21. Scenario 3: impact of parr survival density dependence (decreased rearing habitat quality) on chinook simulation results.

Both R/S and Spawner results were sensitive to the slope of the parr survival curve ($R^2 = 0.75$ and 0.760, respectively; $p < 0.0005$). The mean R/S ratio is an indicator of long-term population survival, because the long-term average R/S needs to be near replacement (1.0) if a population is to persist. Population persistence depends not only on average spawner population size, but also

on how successfully each spawner population's offspring survive to replace their parents. This is a subtle but important point. If the average R/S ratio is close to 1.0, it indicates that even a small average return could be viable; but if it is much less than 1.0 then the offspring aren't surviving at high enough numbers to maintain the population, regardless of the size of average population. The slope of the parr survival curve represents an increase in survival at low populations, thus maintaining replacement R/S ratios over the long-term, which is particularly important under adverse ocean conditions. As the slope declines, this compensatory mechanism declines, and the probability of smaller spawner populations going extinct increases. Hence, both response variables are sensitive to changes in that slope.

Sockeye Analyses

POTENTIAL EFFECTS OF MANAGEMENT ACTIONS: SOCKEYE

Sockeye Scenario 0: Baseline Runs for Lake Billy Chinook

Most of the settings used for the Lake Billy Chinook were determined either by the sockeye workgroup⁶, or personal communication from Chris Kern, who provided summaries from the report on kokanee and sockeye research in Lake Billy Chinook (Thiesfeld et al. 1999). Where no information was available, the Okanogan run reconstruction was used as the starting point. This choice was made because it is believed to be more similar to the Deschutes than other Columbia River populations such as those that rear in Lake Wenatchee, Lake Washington, or Redfish Lake (Mundy 1998–1999). In effect, the Okanogan population was moved to the Deschutes by reducing the number of mainstem dams to two, assuming there were no dams on the Deschutes, and using best available information from Thiesfeld et al (1999) and the sockeye workgroup, to estimate parameters for Lake Billy Chinook. Scenario 0 assumed that:

- Runs were already established in Lake Billy Chinook at recent Okanogan levels: the initial spawner escapement was supplemented, if necessary, in the first three years by sampling from a distribution defined by the maximum (129,600), average (34, 852), and minimum (1,665) escapements to the Okanogan spawners (STREAMNET 1999).
- The initial time period coincided with the time period used for the Okanogan run reconstruction (i.e., the ALPI cycle started at point 50). The simulation ran 50 years.

⁶ As noted earlier, this group consisted of: Phil Mundy (independent consultant, Fisheries and Aquatic Sciences), Don Ratliff (fisheries biologist, PGE), Steve Thiesfeld (fisheries biologist, ODFW); later replaced by Chris Kern (fisheries biologist, ODFW), and Mike Gauvin (fisheries biologist, CTWS).

- Juvenile collection efficacy was 100%.
- Reservoir capacity was defined according to what is known about Lake Billy Chinook (Table 6; see Appendix B).
- The mortalities were lumped into one mortality from egg to fall fry because Lake Billy Chinook data were available as a single stage but not step-by-step (Kern 1999). This means that egg-to-emergence, emergence-to-onset-of-feeding, predation in tributary, and summer mortalities (birth to fall fry, in other words) were all lumped together. Prespawning mortality was still kept separate, and over-winter mortality, when some density-dependent mortality could occur, was also kept separate.

Table 6. Density dependence settings for Lake Billy Chinook sockeye fry.

Parameter	Setting, Lake Billy Chinook (Okanogan)	Explanation
Fall fry mass/length, g/mm	0.11 (0.14)	Kern 1999
Minimum fall fry length, mm	76.0 (80)	Kern 1999
Maximum fall fry length, mm	171.0 (120)	Kern 1999
Minimum reservoir biomass capacity (kg of sockeye + kokanee)	2,044 (24,200)	Arbitrary: 10% of maximum
Maximum reservoir biomass capacity (kg of sockeye + kokanee)	20,439 (242,000)	Thiesfeld et al. 1999
Minimum kokanee population	0 (0)	If all are sockeye, none would be kokanee
Maximum kokanee population	1,824,920 (0)	Assuming all <i>O. nerka</i> are kokanee (Chilcote 1997)

As a rough check on how valid the survival settings might be, survivals by stage were compared against those used for the Kenai and Okanogan run reconstructions and in the literature (sources were Fryer and Mundy 1993, Mullan et al. 1992, and Foerster 1968) (Table 7). The smolt-to-adult survival is a little high for Lake Billy Chinook compared to Fryer’s and Mullan’s estimates for the Columbia, but the egg-to-smolt survival is a little low. Nominal R/S is higher than the simulations produce because this summary ignores stochasticity, variable ocean conditions, and density dependence.

Table 7. Summary of net survival by stage, for Kenai, Okanogan, and Lake Billy Chinook.

Stage	Kenai (Foerster)	Okanogan	Lake Billy Chinook	Published range
Total survival per generation	0.012 (0.010)	0.001	0.004	N/A
Smolt-to-adult	0.047 (0.095)	0.017	0.059	0.011–0.03
Egg-to-smolt	0.025 (0.113)	0.066	0.018	0.024–0.38
Nominal R/S	19.15 (15.99)	1.26	1.32	N/A

The results, which have been log transformed because of the large magnitudes, are shown in Figure 22. Because so many parameters have to be set by judgment rather than data, several trial-and-error simulations were executed in order to try to produce results that were consistent with available data for Lake Billy Chinook. Settings are listed in Appendix B.

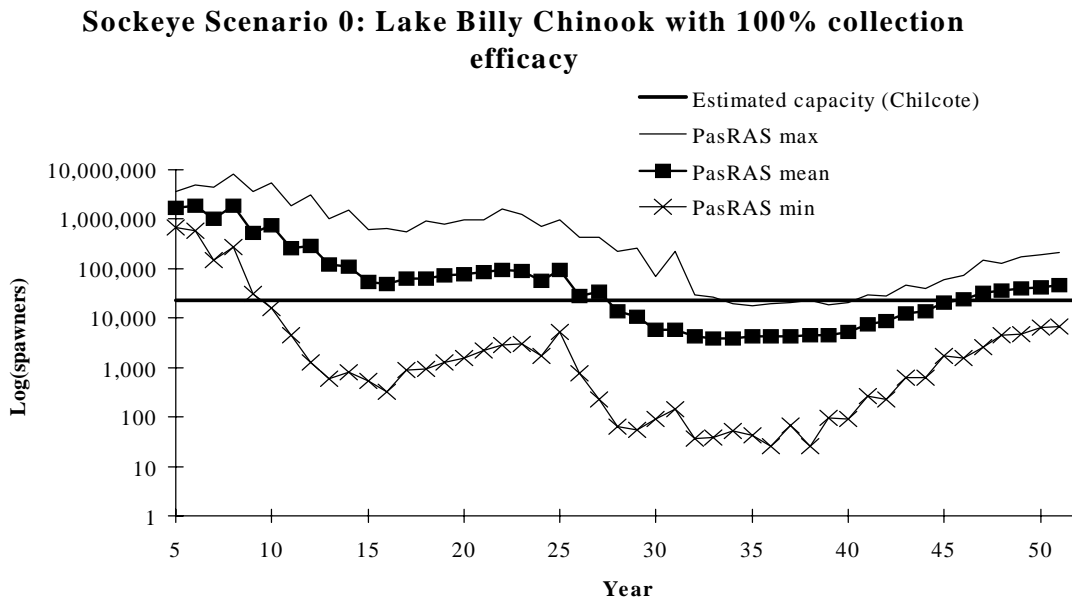


Figure 22. Sockeye Scenario 0: Population performance for Lake Billy Chinook, using settings defined by FTS and sockeye workgroup, with 100% collection efficacy. The flat line graph shows Lake Billy Chinook adult sockeye capacity estimated by Chilcote (1997).

In sockeye Scenario 0, there were no extinctions in 500 replications of 50 years. The mean escapement over 50 years was 209,476 spawners (s.d. = 474,395). The arithmetic mean of recruits per spawner was 1.05, indicating a sustainable run. Figure 22 shows a flat line at 22,342 spawners, which was Chilcote's (1997) estimate of *O. nerka* capacity for Lake Billy Chinook. The simulation hovered nicely around Chilcote's estimate.

The standard deviation (474,395) indicates that there was a sizable stochasticity in the simulation, which was due to the fact that some of the variance is expressed in *PasRAS* as coefficients of variation (C.V.s)⁷.

Sockeye Scenario 1: Effects of Reduced Collection Efficacy.

In Scenario 1, juvenile collection efficacy was reduced by assuming that 70% of the smolts could be attracted to the forebay, 90% of those collected, and 95% of those successfully transported (net efficacy = 0.6). Estimates in the Columbia range from 20% to 60% juvenile collection and transport efficacy, though there is not much data on immediate or delayed juvenile mortality and NMFS requires 95% (Tappel, pers. comm.).

This led to a mean average escapement of 68,731 spawners (s.d. = 215,182), and an arithmetic mean⁸ recruits per spawner of 0.92 (Figure 23). Reducing collection efficacy from 100% to 60% not only reduced the average spawner population by 67% (significant at $p \leq 0.05$); it also led to a 1% probability of extinction within about 30 years.

⁷ Increasing replications to 1,000 reduces the standard deviation by 27%, for example. More replications decreases the confidence interval for an estimate of the mean, but it does not change the graphs of spawners nor R/S noticeably, and so, because each simulation takes quite awhile to execute, 500 replications were used for the report. In a Monte Carlo simulation, the variance of the cumulative mean is often used as an indicator of how many replications are required. For the purposes of this report, 500 replications seemed adequate.

⁸ The geometric mean would be zero whenever any extinctions occurred, because the number of recruits would be zero for one or more years, and a geometric mean calculation that includes a zero value produces a geometric mean of zero. Geometric means are preferred to arithmetic means for probabilities because they give more realistic weight to lower probabilities (Ferguson and Takane 1989). Arithmetic means tend to be higher than geometric means.

Sockeye Scenario 1: Lake Billy Chinook with 60% collection efficacy

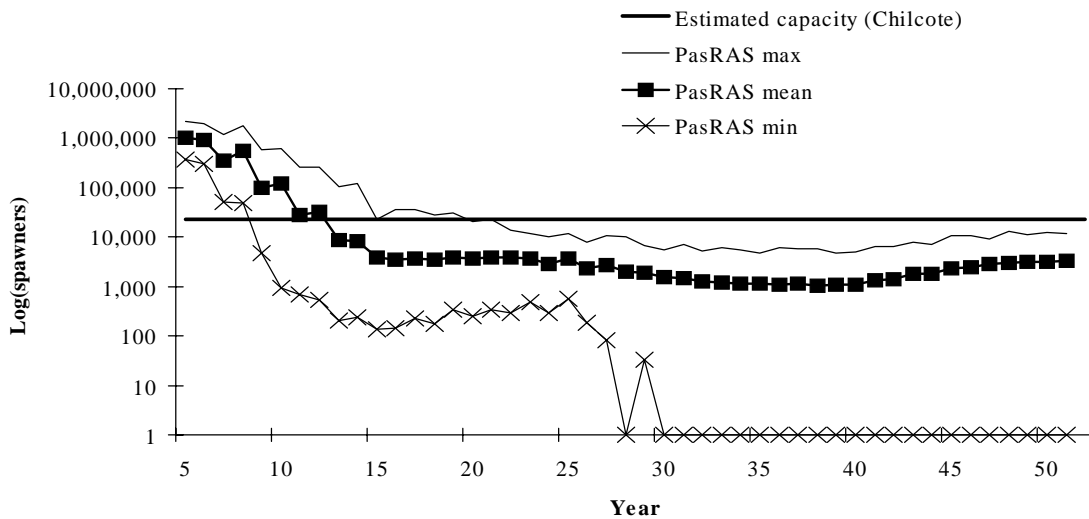


Figure 23. Sockeye Scenario 1: Population performance for Lake Billy Chinook, using settings defined by FTS and sockeye workgroup, with juvenile collection efficacy of 60%.

Sockeye Scenario 2: Effects of Initialization

Sockeye Scenario 2 is the same as Scenario 1 except that instead of an initial escapement for the first three “starter” years based on established Okanogan runs, the initial escapement was assumed to be 2,000 to 8,000 spawners. The results are shown in Figure 24.

Scenario 2 produced somewhat worse results than Scenario 1. Like Scenario 1, the chance of extinction within 50 years was 1%. The mean annual escapement was 17,472 spawners (s.d. = 49,692, a decrease from Scenario 2 of 75% (significant at $p \leq 0.05$). The arithmetic mean recruits per spawner of 0.947.

Sockeye Scenario 2 Lake Billy Chinook with 60% collection efficacy and reduced initial runs

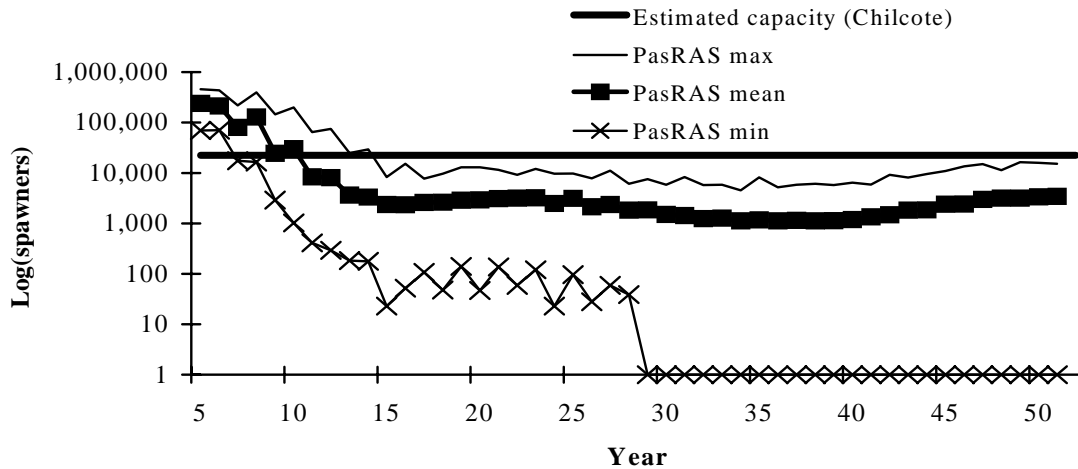


Figure 24. Sockeye Scenario 2: Population performance for Lake Billy Chinook, using settings defined by FTS and sockeye workgroup, with reduced juvenile collection and transport efficacy of 60%, and lower initial escapements.

Sockeye Scenario 3: Effects of Fry Capacity

Scenario 3 was the same as Scenario 2, but with improved rearing habitat quality. There are several ways to simulate improved rearing habitat quality in the simulation. Rearing habitat quality can be increased by increasing carrying capacity, increasing the slope of the over-winter survival curve, or increasing the over-winter survival curve constant (equivalent to assuming higher survival at higher population densities). Both the slope and the constant of the over-winter density-dependent fry survival curve are included in the multivariate sensitivity analyses discussed below, and so for this scenario, rearing habitat quality was improved by increasing the maximum *O. nerka* capacity of the reservoir by 20%, from 10,385 kg to 12,462 kg. The result is shown in Figure 25.

Sockeye Scenario 3
Lake Billy Chinook with 60% collection efficacy, reduced initial runs, and 20% increased fry capacity

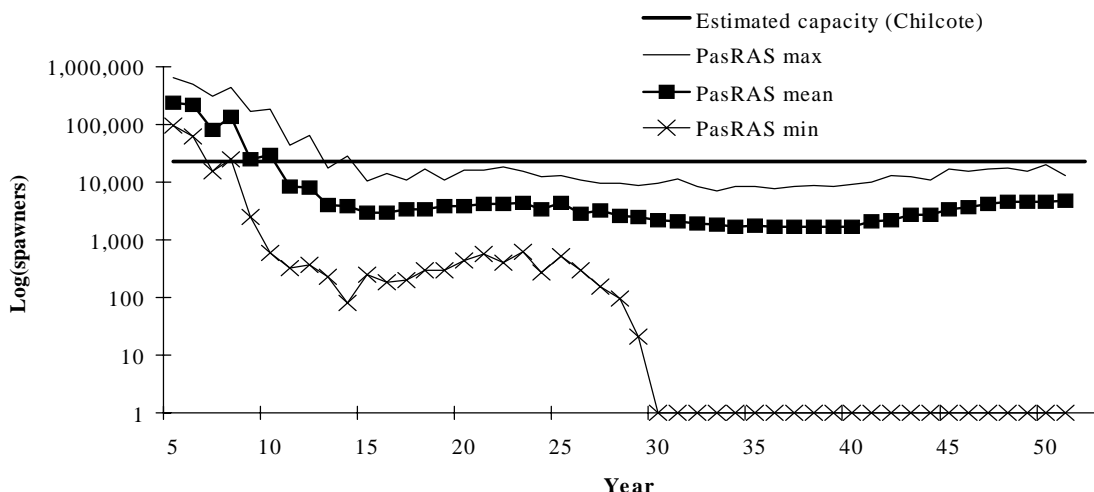


Figure 25. Sockeye Scenario 3: Population performance for Lake Billy Chinook, using settings defined by FTS and sockeye workgroup, with reduced initial runs, transport efficacy of 60%, and improved rearing habitat quality.

Scenario 3 still led to a small (0.4%) chance of extinction, also within 30 years. The mean annual escapement was 18,350 spawners (s.d. = 50,214), with an arithmetic mean of 0.21 recruits per spawner. The 20% increase in over-winter fry capacity led to about a 5% increase in average spawner population size, but the difference was only significant at $p = 0.08$.

Summary of Sockeye Scenario Results

A comparison of the results from the above scenarios is shown in Table 8.

Table 8. Summary of sockeye scenario results.

Scenario	Average spawner population (s.d.)	Arithmetic mean R/S (500 replications)	Probability of extinction (50 years)
0. 100% collection efficacy, start at full seeding, supplement as needed for 3 years	209,476 (474,395)	1.053	0
1. 60% collection efficacy, start at full seeding, supplement as needed for 3 years	68,731 (215,182)	0.919	1%
2. 60% efficacy, start with 1,000 to 3,000 spawners and supplement as needed for 3 years	17,472 (49,692)	0.947	1%
3. 60% efficacy, start with 1,000 to 3,000 spawners and supplement as needed for 3 years, but increase smolt capacity by 20%	18,350 (50,214)	0.966	0.4%

Because extinctions happened in all the scenarios simulated, the geometric mean R/S was 0 for all scenarios, so arithmetic means are shown instead. Only one of the mean R/S ratios was in the range considered sustainable—the one with 100% collection efficacy—and the other simulations led fairly quickly to extinction.

Because these scenarios tended to be disappointingly pessimistic, it is important to recognize that they were based on extremely limited information. It is quite possible, in fact more than likely, that the assumptions about and the interactions between kokanee, bull trout, and sockeye are wrong. For these reasons, the results summarized in Table 8 should not be used for purposes beyond which they could be considered valid. The first scenario, which simulated essentially a no-dam scenario (100% collection and transport efficacy), matched Chilcote’s carrying capacity estimates reasonably well and produced sustainable populations even without supplementation; but increasing mortality at this stage led to some extinctions. Collection efficacy, or presumably other mortality at the fry-to-smolt stage, obviously has a big impact in the simulations. However, the multivariate sensitivity analysis is a much more relevant approach to evaluating what might have the biggest effect on achieving sustainable populations.

SOCKEYE SENSITIVITY ANALYSIS

Many preliminary ad-hoc analyses were conducted in order to reduce the list of candidate parameters, to determine how wide the parameter ranges needed to be, and to determine the most useful response variable for the sockeye sensitivity analyses. As was the case for chinook, validation and run reconstruction exercises indicated that all of the parameters associated with ocean mortalities (i.e., annual mortality by age of fish, mortality as a function of the ALPI, size assumed for each age of fish, and age structuring) are critically important. This is because the

ocean is where much of the mortality occurs in the real world, and because of the high levels of stochasticity and unpredictability in the “ocean” within the model. Because these ocean parameters dominate any sensitivity analysis, in order to conduct a multivariate sensitivity analysis on other parameters, stochasticity was minimized as much as possible for the ocean parameters. This meant ignoring ALPI cyclic impacts on mortality, and setting C.V.s = 0 for annual mortality. This still allowed considerable stochasticity due to age structuring, fecundity, and other parameters, while still allowing Analysis of Variances to produce acceptable significance levels, with 1,000 replications. Because it turned out to be difficult to not have most or all the end in extinction, these replications were only run to 25 years.

The parameter settings used are shown in Table 9. In order to try to avoid skewing the results by making the parameter ranges larger for one parameter than for others, the percentage difference between the “MIN” and “MAX” levels was set at a uniform 20%. All the other settings were set to the Lake Billy Chinook settings (Appendix B), but with net weighted ocean mortality between 0.9 (large smolts) and 0.94 (small smolts) rather than depending on the ALPI. The response variable that turned out to produce significant results was the geometric mean R/S (no doubt because the geometric mean is more sensitive to extinctions than the average spawner population size).

Table 9. Parameter ranges for multivariate sockeye sensitivity analyses.

Parameter	Min	Max	Max/Min
Juvenile collection efficacy	0.8	1	1.25
Over-winter survival at K	0.32	0.4	1.25
Spawner Success	0.76	0.95	1.25
Egg-to-fall fry mortality	0.76	0.95	1.25
Juv. Columbia (2 dams)	0.250	0.313	1.25
Adult transport	0.010	0.013	1.25
Density dependence	0.16	0.2	1.25
Reservoir carrying capacity	18,395	20,439	1.25

The results are shown in Table 10. Analysis of Variance was done by conducting a regression on variables that had been transformed to the standardized form as discussed above (Ferguson and Takane 1989). As required by this method, parameters were shown to be orthogonal by calculating a correlation matrix, in which the highest magnitude correlation was 0.05. Analysis of Variances were conducted for two different response variables: average spawner population size, and geometric mean R/S. Although both regressions were significant at $p < 0.00005$, only

the geometric mean R/S Analysis of Variance produced acceptable R_{adj}^2 ($R_{adj}^2=0.16$ for spawners, 0.97 for geometric mean R/S).

Aside from the ocean parameters, the significant parameters for the sockeye simulation were found to be (in order of effect), egg-to-fall fry mortality, juvenile collection and transport efficacy, spawner success, over-winter minimum survival. Not found to be significant (at $p \leq 0.05$) were adult losses migrating through the Columbia (due to Bonneville and The Dalles dams), reservoir carrying capacity, and the slope of the density-dependent over-winter fry survival curve (Table 10). “Not significant” does not mean the mortalities are not significant to juvenile sockeye—it only means that other parameters contributed so much more to the variance of the response variable (geometric mean R/S) that the contributions of the “not significant” parameters were not distinguishable.

Table 10. Analysis of Variance for multivariate sockeye sensitivity analysis.

Geometric mean R/S	Coefficients	P-value	Contribution to variance
Egg-to-fall fry mortality	-0.960	0.000	0.922
Juvenile collection efficacy	0.103	0.000	0.011
Spawner Success	0.102	0.000	0.010
Over-winter survival at K	0.097	0.000	0.009
Juv. Columbia (2 dams)	-0.038	0.000	0.001
Adult transport	-0.009	0.118	0.000
Density dependence	-0.007	0.251	0.000
Reservoir carrying capacity	-0.001	0.901	0.000

Table 10 also lists the rank-order contributions to variance of the significant parameters, which indicates their relative impacts on the outcome. Of the outcome variance, 92% is explained by the analysis ($R_{adj}^2 = 0.97$). The “contribution to variance” column in Table 10 indicates that most—92%—of that is explained by egg-to-fall fry mortality. *PasRAS* provides the capability of representing mortalities through this stage in some detail, but data from Lake Billy Chinook were only available as a single “chunk” from egg-to-fall fry (Kern 1999). The most obvious explanation of this result is that the way this stage is represented covers a large piece of time

(egg to fall fry), and during this stage, the population numbers are so high, and mortality is so high, that the fractional impacts at subsequent stages have little effect on the outcome. Type II error was found to be less than 0.0005 (Montgomery 1985). This means that the probability of incorrectly failing to reject the null hypothesis (i.e., the probability of concluding that one of the parameters has negligible effect on recruits per spawner—and being wrong) is less than 0.0005.

Egg-to-fall fry mortality—mortality in the tributaries—is an indicator of tributary habitat quality and impacts from predation in the tributaries. Spawner success is also an indicator of tributary habitat quality. Over-winter mortality is an indicator of rearing (reservoir) habitat quality. Thus, the most influential freshwater parameter for the sockeye simulation model were found to be:

1. Egg-to-fall-fry mortality
2. Juvenile collection efficacy
3. Spawner success
4. Reservoir habitat quality

Aside from the freshwater parameters, the impact of ocean mortality parameters (mortality by age, age structuring, fecundity and length as functions of age) is extremely important, and is the major source of stochasticity and hence, extinctions, in the simulations.

In addition to the multivariate sensitivity analysis, a single-parameter analyses for juvenile passage efficacy was conducted for sockeye. The FTS was originally formed to evaluate the feasibility of fish passage, and so one of the primary questions was: “how good might collection efficacy have to be?” There was thus particular interest in how the results of the model vary with juvenile collection efficacy.

Sockeye Single Parameter Sensitivity: Effects of Collection Efficacy.

The simulation was run with the same settings as were used for scenario 1, except that juvenile collection efficacy was varied, using the sensitivity analysis option for sampling from a uniform distribution. The uniform distribution was set to a range of 0.6 to 1.0. One thousand replications were conducted, of 50 years each. The results are shown in Figure 26.

Impact of Efficacy on sockeye results

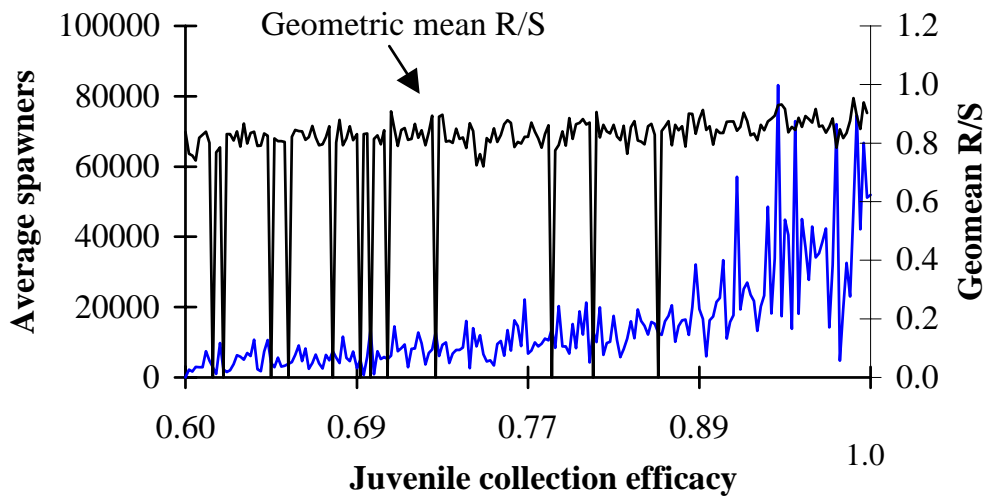


Figure 26. Impact of juvenile collection efficacy on sockeye results.

Juvenile collection efficacies below about 0.9 produce geometric mean R/S ratios that are not sustainable and/or led to extinctions. This is illustrated by the way the plot oscillates between 0 and about 0.9 R/S, up to what appears to be a threshold at 0.85. Above this threshold, the geometric mean increases linearly with collection efficacy (though the scale on the graph does not show it very well, the geometric mean does increase). There is a similar threshold for the 50th-year spawner populations, no doubt for the same reason: populations tend to go extinct within 50 years if collection efficacy is less than approximately 0.9.

Although all of the sockeye scenarios achieved geometric mean R/S ratios that are marginally sustainable (greater than 0.9), it is clear that as collection efficacy goes up, so does the geometric mean R/S ratio. Other than the oscillations due to extinction rates, the geometric mean increases linearly with collection efficacy. The 25th-year spawner population size increases more exponentially with efficacy.

RESPONSES TO VALIDATION QUESTIONS

Returning to the questions that were listed above as validation criteria, here is a summary of *PasRAS* validation conclusions:

1. What characteristics of salmon life cycles have been left out or simplified, and what effects might those omissions or simplifications have on model output?

The FTS and informal reviewers were asked to brainstorm a list of missing or over-simplified characteristics in the *PasRAS* model of both species. Here is a summary of the most significant items on that list:

- Independence of stages. Although density-dependent survival mechanisms prevent both species models from being linear, there are many linear assumptions built in that are probably not linear in the real world. A “linear” model here means that each stage is additive and independent from previous stages. For example, the Monte Carlo mechanisms in the model treat mortalities from one stage to the next as though they are independent, when it may be that high mortality at one stage may be correlated with high mortality at the next, due, for example, to a severe winter or unusually dry year. Conditions at one stage may affect survival at later stages, as for example the condition of smolts may make them more vulnerable to predation in the estuary and ocean, or non-lethal impacts at one stage causing uncouneted mortalities at later stages. Such delayed mortalities can be modeled in the estuary part of the model, but mortalities at multiple stages cannot currently be linked to co-vary as they might do in nature.
- Environmental trends. Arguably the most significant trend pattern, ocean mortality, is represented by simulating the Aleutian Low Pressure Index, which is believed to play a significant role in ocean survival of Oregon salmon. Freshwater and marine stochasticity and catastrophic events (disease and floods) are included, though how realistic they are for these populations and at this time is not known. Long-term freshwater habitat quality degradation or improvement trends are also not included. The impact of leaving out these trending capabilities depends on whether the constant (though far from uniform) freshwater mortality settings in *PasRAS* are worse or better than the net effect of actual trends would be. Freshwater habitat quality trends have tended to be more downward than upward throughout the Columbia basin in recent decades. Ongoing efforts towards improvement will not likely show effects for some years. If the most likely trend for freshwater habitat quality is a continued downward slope, then *PasRAS* most likely underestimates freshwater mortality.

- Missing or simplified interactions. Interactions among year classes and between species are either missing or greatly simplified in both models. One potentially important phenomenon is the multi-year impacts of previous returns on current-year productivity, for sockeye in the reservoir and chinook in the tributaries (Kline et al. 1993, Mathisen 1972, Piorkowski 1997). Because zooplankton life cycles run one to two years, large sockeye returns one year may deplete the food supply for a year or two following, and conversely for small returns, leading to a classic predator-prey, cyclic pattern over short time periods. Large returns one year could also increase productivity in following years, particularly for spring chinook, if the carcasses of previous years' spawners fertilized the spawning beds and tributary rearing areas (Bilby et al. 1996, Finney 1998, Kline et al. 1993, Mathisen 1972, Piorkowski 1997, Thornton 1990).

Such delayed density dependence is not included at all in the chinook version. It is included in one version of the sockeye model, though in a highly speculative way because so little is known about what the actual parameters and relationships might be. Thus, it is difficult to say how faithfully the interaction is modeled, and it was not used for this report. Run reconstructions for the Kenai and Okanogan could be produced more accurately with the speculative mechanisms included than by using a classic linear model with no density dependence or feedback loops; but so many of the parameters had to be set by trial-and-error that it is hard to say how realistic the simulated mechanisms are.

Also speculatively modeled are interactions between sockeye and kokanee, which would further complicate the multi-year feedback loops involving zooplankton. Kokanee populations are modeled as though they vary randomly, when in reality they probably interact in complex ways with zooplankton and bull trout now, and would with sockeye in the future.

Finally, although the sockeye version allowed carrying capacity to vary randomly over a user-defined range, such variation was missing in the chinook model. In the chinook version, reach chinook smolt capacities were defined as constants. In the sockeye version, reservoir sockeye capacity was defined in a complicated way that varies with kokanee populations, which vary randomly. There is thus quite a bit more "noise" in the modeled sockeye carrying capacity.

In contrast to the sockeye version, the effect of treating chinook reach capacities as constant would be to reduce the stochasticity of the density-dependent parr survival mechanism, which, insofar as extinction rates go up with stochasticity, could reduce the simulated extinction rate.

- Simplistic responses to ocean conditions. *PasRAS* has an option to allow chinook or sockeye salmon to return to freshwater sooner, if ocean conditions are good, than they would if ocean conditions are bad. This is to simulate the assumption that fish would be able to mature faster under good ocean conditions. If they return sooner, then the fish will be smaller and the fecundity lower, but their numbers will be higher because they were not subjected to an additional year's mortality. There appears to be an environmentally mediated adaptive pattern to the age structuring observed in some returning spawner populations, in addition to a genetically coded element. For example, the relatively high frequency of three-year-old spawners in the Okanogan may be such an adaptive response (Mundy 1998–1999). This option had a deleterious effect on run reconstructions for both sockeye and chinook and thus was not included in any of the scenarios discussed in this report.

The results may be a consequence of the way the option was implemented. The cutoff points are defined in the code according to the total ocean mortality each year, to cover the range of total ocean mortality that seemed consistent with expert input (Lawson 1998, Mundy 1998–1999, Schreck 1998–1999). A better way to implement this option would be to allow the user to define the cutoff points and age structuring shifts, rather than having it hard-wired in the code as it is now. Regardless, it is difficult to know how reasonable the current assumptions might be. Turning this option off leads to lower population sizes and a slightly increased chance of extinction, all other settings being equal.

Another difficulty with responses to ocean conditions is the way the ocean fishery—and, for that matter, the freshwater fishery—is implemented. Newer versions of *PasRAS* allow several management decision rules to be simulated, but the versions used by the FTS assumed harvest to be set as constant annual rates, when in reality they vary widely in response to management decision-making processes. A crude option that allows ocean fishery mortality to co-vary with ocean mortality does introduce more realistic cyclic behavior, but assumptions about management decision-making need to be further examined.

The impact of these limitations is primarily to reduce stochasticity in *PasRAS* results, which would tend to reduce extinction probabilities, all other things being equal. However, assuming a constant harvest rate in the ocean neglects the flexibility managers have to close down a fishery when populations decline, perhaps preventing extinctions that *PasRAS* would allow to occur. On the other hand, assuming a *low* mean harvest rate in the ocean neglects the possibility of occasional high harvest levels such as those observed in data sets—hence potentially leading to extinctions that *PasRAS* would not produce.

- Missing or inadequately modeled life history strategies.

Only one life history strategy for sockeye is modeled⁹, and there are two life history strategy options for spring chinook (stream-type and ocean-type). The ocean-type life history for chinook does not adjust age nor size of smolts to account for the likelihood that ocean-type smolts would be smaller on arrival in the ocean. Although the stream-type strategy for chinook is dominant at this time in the Deschutes, the ocean-type life history does exist below Sherars Falls. Life history diversity is an important component of species resilience, and inadequately representing this diversity would lead to increased probability of simulated extinction.

An important life history poorly modeled in the sockeye version is the kokanee life history type, which is currently well-established in Lake Billy Chinook. Interactions among juvenile sockeye, resident kokanee, and their common prey base (reservoir zooplankton), could be fundamentally important, particularly over 1–3 year time periods. They are included in *PasRAS* at this time but in a speculative way. The potential effects of these interactions, or errors in the way they are represented, are unknown.

- Metapopulation dynamics. Straying is simulated among the tributaries to Lake Billy Chinook, but not between the Deschutes below the dams and tributaries above the dams, nor between any other Columbia basins and the Deschutes. Deschutes spring chinook are in the Mid-Columbia ESU, and it is not known which ESU Deschutes sockeye would be in. Because the out-of-basin straying rate has occasionally been high for hatchery chinook on the Deschutes (Fagan 1998), and because much of this straying is from other ESUs, this may be an important missing feature. It is unknown at what rate Deschutes chinook stray out-of-basin. Out of basin strays may help maintain dwindling runs by recolonization, or they may threaten runs via disease or genetic impacts.
- Density dependence. Performance of both models is strongly affected by the mechanisms associated with density dependence. Three key concerns about the approaches used are that (1) density dependence is only imposed at the parr-to-smolt stage for chinook, and at the over-winter fry-to-smolt stage for sockeye, whereas density dependence may be important at other stages and may even not be important at the stages selected; (2) the mathematical function used is based on what Nickelson and Lawson (1998) used for coho, and may not be appropriate for chinook nor sockeye; and (3) there is no quality information readily available to support the parameter settings used for the slope and constant of the functions. It is hoped that including density dependence at one life history stage could at

⁹ Kokanee are only included as a randomly varying element affecting reservoir carrying capacity for sockeye.

least partially account for density-dependent mechanisms missing at other stages. The mathematical function used and the parameter settings are certainly questionable. Although it seems reasonable to expect that the overall shape of a density-dependent survival function would exhibit a declining slope, it is possible that there is a threshold mechanism that should have been represented. Other potentially density-dependent phenomena, such as superimposition of redds by one species on another, or increased probability of some disease risks, are simply ignored.

Although the sockeye density dependence is more sophisticated and noisy than the chinook version, it is also very speculative. The chinook version of density dependence has been used in other models [e.g, Nickelson and Lawson's (1998) coho model], whereas the sockeye version has not. However, there is little doubt that the chinook density dependence mechanism is a potentially important simplification because carrying capacity depends on many factors not included in a relatively simple model like *PasRAS*.

2. Is the structure of PasRAS adequate to serve the purposes for which it will be used?

Yes, to the extent that the baseline scenarios, the sensitivity analyses, and the management impact scenarios produce results that are consistent with reviewers' and current users' expectations. If the intent were to use it to estimate how big populations would be and what the probability of extinction would be for chinook or sockeye reintroduction efforts, the answer would obviously be "no."

PasRAS is intended to be used as a "thinking" tool, for helping people reach agreement on a common technical understanding of sockeye and spring chinook salmon life cycles and the risks associated with fish passage. It is intended to help people think through possible scenarios, and reach agreement on the relative importance of risks, possible impacts of management activities, and prioritization of research and habitat improvement efforts. Although *PasRAS* is not intended to be used for predictive purposes, the fact that it is generally consistent with the longer-term patterns for spring chinook runs at Lower Granite Dam on the Snake (1975–1995), spring chinook runs above Sherars Falls on the Deschutes (1977–1998), sockeye runs on the Kenai (1969–1990), and sockeye runs on Lake Okanogan (1960–1996), helps build confidence that the key variables are captured well enough to use it for the intended purposes.

3. Are the parameters and ranges used valid?

Yes, to the extent that the model produces reasonable output for a wide range of scenarios. The parameters used so far are the best that were readily available. Most of what is known about sockeye salmon today is known about Alaska and Canadian populations, which undoubtedly are

different from what Lake Billy Chinook populations might become. Jeff Fryer's 1995 dissertation was invaluable for developing reasonable scenarios for Columbia River sockeye, and the longer-term data sets downloaded from STREAMNET (<http://www.streamnet.org>) were also useful. Thiesfeld et al.'s (1999) study of Lake Billy Chinook kokanee and their implications for sockeye reintroduction was the only relevant local data available.

The best available regional data at this time that could be applied to Columbia Basin spring chinook is from the PATH team. The best available *local* data for spring chinook is from ODFW's most recent report on the subject (Lindsay et al. 1989), Mullan et al.'s (1992) study of Columbia River spring chinook, and unpublished data from the CTWS (Fagan 1998). Most valuable for the Lake Billy Chinook simulations was the *Habrate* database developed by Burke and Dambacher (ODFW 1999), as adjusted by Mike Riehle (USFS).

As for the settings used for the validation and sensitivity analyses, recommendations were provided by Dr. Mundy (sockeye) and Dr. Schreck (chinook). Dr. Mundy based his recommendations primarily on Foerster's work with Alaska sockeye (Foerster 1968) and on Fryer's dissertation (Fryer 1995), and Dr. Schreck relied primarily on data developed by the PATH team. For spring chinook, parameters for which appropriate settings were not known were calibrated by trial and error, using known run sizes at Lower Granite Dam on the Snake, and above Sherars Falls on the Deschutes. For sockeye, calibration by trial and error was developed using known run sizes on the Kenai in Alaska, and the Okanogan sockeye runs on the upper Columbia in Washington state.

As more applicable data become available from current habitat analysis and studies in the Deschutes basin, Columbia system, and the ocean, the parameter settings should be revised to better represent the actual system of interest.

4. Does PasRAS produce long-term steady-state sustainability when it should, does it produce declines when it should, and does it produce extinctions when it should?

The validation procedures described above demonstrated that *PasRAS* produced long-term sustainability, declines, and extinctions for both sockeye and chinook in ways that seem reasonable. It does an acceptable job when used to simulate existing sockeye and spring chinook runs. Longer-term cyclic patterns are consistent with actual patterns, and although shorter-term boom-and-bust cycles are evident in single replications, they "average out" as would be expected in the plots of simulation means. However, because it is virtually impossible to produce extinctions with the chinook species model unless density dependence is decreased, and because it is difficult *not* to produce extinctions with the sockeye model without producing ridiculously large run sizes, the results for the probability of extinction are questionable at best.

Short-term, as well as long-term, population swings are familiar patterns in actual salmon populations. Although it is reassuring to see that *PasRAS* can produce familiar patterns, it is impossible to say whether the simulated patterns are due to faithful representation of real mechanisms or mechanical artifacts built into the computer code. The computer code reproduces the FTS' and consulting experts' best judgments of how to simulate highly complex, nonlinear systems. However, reducing such complex systems to a sequential life history model involves many simplifications that no doubt reduce the accuracy of the performance. Nonetheless, because extinctions in the real world tend to occur in stochastic "bust" years as populations generally decline, the fact that *PasRAS* does exhibit such boom and bust stochasticity, and associated extinctions, lends credence to the qualitative results.

5. What are the anomalous or surprising behaviors that have been observed, and have they all been explained and/or corrected?

At this time, the anomalous or surprising behaviors that are known, and have been caused by code bugs, have all been corrected. Perhaps the most striking anomalous behaviors occurred when older versions of *PasRAS* output was compared to actual escapement data for the Kenai and Okanogan sockeye populations. Those anomalies consisted of short-term (1–3 years) boom-and-bust cyclic patterns in real populations that older versions of *PasRAS* did not reproduce because mechanisms that would produce that kind of cyclic patterns, such as productivity feedback loops and predator-prey interactions, were not well enough represented in the model. When the computer model was modified in order to include hypothesized mechanisms to represent such interactions, short-term cyclic patterns did appear in single *PasRAS* replicates.

Another anomaly was the high peak spring chinook escapement in 1977–1978 at Lower Granite on the Snake River, which did not show up in the Deschutes counts at Sherars Falls nor in *PasRAS*. The Snake River peaks were probably due to the fact that they were the last year classes to return from brood years prior to completion of the last Snake River dams.

Habrate also produced some surprises because of the few reaches that ended up supporting chinook—not only how few there were, but which ones they were. In particular, none of the Metolius reaches ended up with chinook populations. Mike Riehle, who adapted *Habrate* for these simulations, explained that the reason for this result is that under the rules of the *Habrate* habitat rating system [developed by Burke and Dambacher (1999)], chinook rearing only occurs in pool habitat. The USFS protocol only considered pools if they were as wide as they were long. Short pools are common in broad, spring-fed channels of the Metolius system, but these pool habitats or side pool habitats in the Metolius were not included in the total percentage of pools, and may have resulted in a lower rating of habitat quality for the Metolius River. Also, older surveys conducted by the USFS and current ODFW protocols inventory pools and glides separately. In 4th and 5th order stream channels with good depth, these glides may provide

some habitat for rearing chinook. Deep glide habitats were not included in the percent of pool habitat in *Habrata* and may have lowered the habitat quality rating for chinook in some reaches.

6. How do PasRAS structure and behavior compare to similar models? Does PasRAS address limitations in those models, and if so, how? Are limitations in PasRAS revealed by comparison to those models, and if so, how?

Note: this section was provided by Dr. Philip Mundy.

PasRAS is conceptually and structurally analogous to other salmon life cycle models that have been developed within the Columbia River basin. Two analogous models intensively studied during the PATH process are the CriSP Model (James Anderson, University of Washington, Seattle) and the FLUSH model (Howard Schaller, ODFW; Charlie Petrosky, IDFG; Weber, CRITFC, et al.). Both models were initially developed for application to a single species (spring chinook). Both incorporate existing information on mortality by life cycle stage with assumptions regarding the effects of key variables influencing survival, such as river flow and transportation of juveniles around the dams, to estimate population growth. CriSP is extensively stochasticized, whereas FLUSH is predominantly deterministic. Both FLUSH and CriSP have been used extensively to understand the possible effects of changes in the configuration and operation of the federal hydroelectric system on growth of chinook populations. *PasRAS* is analogous to CriSP and FLUSH in that all three are age structured life cycle models of chinook salmon population dynamics using run reconstruction techniques.

A third model analogous to *PasRAS* is the stochastic life cycle model, SLCM, developed by the USFS (Danny Lee, Boise) for understanding the potential effects of key anthropogenic factors on the growth of salmon populations. The SLCM is analogous to *PasRAS* in being a comprehensive, fully stochastic representation of the chinook salmon life cycle with output describing the abundance of the population through time.

PasRAS addresses a limitation of existing Columbia River models by modeling sockeye salmon. Other Columbia River models were limited to salmon with coded wire tag recovery histories, such as chinook, so that comparing model outputs to tag recoveries could validate the model's representation of fisheries effects. *PasRAS* has imposed no such limitation, and as a consequence it has incorporated a structure appropriate to the life history strategies of sockeye salmon, and it has assembled a reasonable set of life cycle parameters from local Columbia River stocks and from Canadian and Alaskan stocks. Fishery effects on Columbia River sockeye in the ocean are thought to be negligible, and the effects of Columbia River fisheries are reasonably well documented without the use of coded wire tags. As far as we know, *PasRAS* is the only sockeye model that models delayed density dependence and variable carrying capacity, and includes kokanee in rearing interactions with sockeye.

Comparison of model outputs among CriSP, FLUSH, SLCM, and *PasRAS* has not been accomplished in a rigorous quantitative manner. Based on a visual comparison of similitude of model outputs to time series of chinook returns, the fidelity of *PasRAS* chinook output to the time series of chinook returns is reasonably similar to that of the other three models. A coastal coho version of *PasRAS*, which is essentially the chinook version with different parameter settings, is currently being evaluated by Tom Nickelson (ODFW).

7. What parameters are *PasRAS* results most sensitive to? How do these sensitivities correspond to what is known about the real world?

The sensitivity analyses indicated that for both species models, any of the parameters associated with ocean mortalities are critically important, both because the ocean is where much of the mortality occurs in the real world, and because of the high levels of stochasticity and unpredictability in the ocean.

Because ocean conditions dominate any sensitivity analysis, in order to conduct a multivariate sensitivity analysis on freshwater parameters, stochasticity was minimized for the ocean parameters. This meant turning off the ALPI option, holding the age structuring constant rather than allowing fish to return sooner if conditions improved, and setting variances = 0 for annual mortality and parr survival. This still allowed considerable stochasticity due to age structuring, fecundity, and ocean mortality, while still producing acceptable Analysis of Variances.

The most influential freshwater parameters for the sockeye simulation model were found to be:

1. Egg-to-fall-fry mortality
2. Juvenile collection efficacy
3. Spawner success
4. Reservoir habitat quality

The most influential freshwater parameters for chinook were found to be:

1. Tributary habitat quality
2. Juvenile collection efficacy and quality (as it affects juvenile mortality, smoltification and stress)
3. Columbia River dam mortalities
4. Mortalities suffered by adults returning up the Deschutes

It is important not to forget that the sensitivity analyses were for *freshwater* parameters. Ocean parameters (mortality by age, age structuring, fecundity and length as functions of age) are responsible for most of the variance in any scenario simulated. It is this stochasticity that leads to higher rates of extinction than would otherwise be indicated in non-stochastic analyses.

These sensitivities are consistent with what is believed about the real world, as evidenced by studies that have been ongoing on the Deschutes (PGE 1997). The critical freshwater parameter values for sockeye are consistent with similar modeling efforts for salmon originating from habitats behind hydroelectric dams (Fryer and Mundy 1993). Collection efficacies and dam mortalities are well known bottlenecks limiting the growth of salmon populations. The same factors are likely to be important for any salmon population originating in dammed habitat (Fryer and Mundy 1993), so it is not surprising that spring chinook and sockeye had similar critical freshwater parameters.

8. How does PasRAS respond to simulated conditions or policy changes? In particular, how does PasRAS respond to changes in marine and freshwater environments; how does PasRAS respond to changes in spawning and rearing habitat quality; and what kind of collection efficacies might be required in order to compensate for ocean mortality and less-than-optimal spawning and rearing habitat quality? Do these responses seem reasonable?

PasRAS responds as might be expected to changes in marine and freshwater environment: simulated populations increase in size, and the probability of extinction declines, when ocean conditions improve or spawning and rearing habitat quality improves, all other things being equal. Regarding specific management options, three are summarized here:

Management option 1: Improve collection efficacy: the more successful the juvenile collection efficacy and quality is in the simulations, the larger the run sizes and the higher the probability the simulated populations would survive. For sockeye as well as chinook, average run sizes increased linearly with increased collection efficacy. Over the long term, reducing efficacy from 100% to 60% reduced the average sockeye spawner population by 67%, and increased the probability of extinction within 50 years from 0% to 1%. It also reduced the arithmetic mean R/S from 1.1 to 0.92, which produces a slowly declining population over 50 years.

Reducing efficacy from 100% to 60% reduced the average chinook spawner population by 56%. Although the probability of extinction was zero either way for chinook, the geometric mean R/S was reduced from 0.91 to 0.89, which is below replacement over the long run. Also of concern

is that the average number of reaches still being used¹⁰ in the 50th year went from 6.8 to 4.5—even under the most optimistic scenarios, out of 85 reaches originally populated, only 6 to 10 reaches were still being used by about the 10th year of these simulations.

Management option 2: Larger initial populations. Long-term simulation behavior was also affected by assumptions about initial population sizes, though not to a great extent. For sockeye, the average population over time was much greater when initial populations were large (209,476 versus 17,472 average escapement for large and small initial populations respectively). Large initial populations were meant to represent how a sockeye run at estimated Lake Billy Chinook capacity might perform once it actually was somehow established in Lake Billy Chinook. Smaller initial populations were meant to represent how a sockeye run might be established in the first place.

For chinook, initial population size had negligible effect on results, most likely because of two mechanisms in the simulation: (1) most of the 92 reaches initially populated by *Habrata* were depopulated within a few years regardless of scenario, leading to less than 10 still populated by the 50th year; and (2) the steepness of the density-dependent parr survival curve meant that parr survival increases dramatically at low population densities, thus counteracting trends towards extinction.

Management option 3: Improved rearing habitat quality. Improving rearing habitat quality for chinook means improving tributary survival, whereas for sockeye it means improving tributary as well as reservoir survival. For chinook this question was investigated by reducing smolt capacity by 20% and comparing the results to the baseline scenario. The effect was almost linear for run sizes: a 20% change in smolt capacity produced a simulated 21% change in spawner population size.

For sockeye, *O. nerka* reservoir carrying capacity, expressed in kilograms of sockeye plus kokanee combined, was increased 20%. The effect of increasing reservoir rearing habitat quality for sockeye was only about a 5% increase in average escapement. This result is partly due to the way egg-to-fry mortality was lumped into such a high rate (95%), which meant that subsequent stages had few fish to kill anyway. It was also partly a threshold phenomenon: when Scenarios 1 and 2 were mistakenly run with half the nominal carrying capacity, extinction rates of 100% were hard to avoid even by decreasing mortalities at other stages.

¹⁰ The chinook simulation defines habitat quality by spawning reach, whereas the sockeye simulation assumes fractions of total habitat. This difference is because chinook rear in the tributaries whereas sockeye rear in the reservoir. 92 reaches are populated at the beginning of the Lake Billy Chinook simulations.

CONCLUSION

PasRAS simulations produced relatively optimistic results for both chinook and sockeye. Although average run sizes were small (less than about 1,200 spawners for any of them, and 220 for the most representative), none of the chinook scenario simulations led to extinctions. Long-term outlook, as represented by the geometric mean R/S, was not quite at long-term replacement (ranging around 0.9). Low ocean survival can lead to extinctions, but ocean survival in the chinook scenarios varied from 2.3% to 4.0% and thus was not particularly benign. The big question affecting chinook *PasRAS* validity is what the parameters should be for the parr survival curve, because the results are highly sensitive to the level of density-dependent survival represented by that curve. Local data that were available (Lindsay et al. 1989) had to be modified so much in order to use them that the results were questionable. The curve used to produce satisfactory run reconstructions for Snake and Deschutes spring chinook populations was nonetheless based more on judgment and hearsay than real data (Schreck 1998–1999).

For sockeye, *PasRAS* simulations produced similarly encouraging results. Simulated scenarios indicate that robust populations of sockeye, once established, would still be marginally robust, if juvenile collection efficacies were extremely high (above about 85%) compared to what they have been on the Columbia. When parameter settings were assumed to be more consistent with the risks sockeye populations in the Deschutes would actually face (collection efficacies of 0.6, initial populations of 2,000–8,000 spawners), *PasRAS* simulations produced lower run sizes and small probabilities of extinction within 50 years. Because so many assumptions were required to produce the Lake Billy Chinook sockeye simulations, it would not be appropriate to consider these results predictive in any way. The simulations depended on calibrations needed to produce a satisfactory Okanogan run reconstruction, limited data from Alaska (Foerster 1968) and the Okanogan (Fryer 1995), and highly speculative judgments about Lake Billy Chinook sockeye capacity provided by Chilcote (1997), Phil Mundy, and members of the FTS. Instead, the sockeye simulations should be considered a best-guess baseline that could be used for the sensitivity analyses and for evaluating management options.

Answering the final question, about how high collection efficacies might need to be to maintain, let alone initially establish, sustainable runs of spring chinook and sockeye above the PRB project, the simulations appear to suggest that they would need to be better than they seem to be in the Columbia. This is not unreasonable, given the trends of Columbia salmon stocks, including lower river spring chinook population in the Warm Springs River. Chinook simulations produced more optimistic results than sockeye simulations. The fact that there are also better data available for chinook than for sockeye could imply that sockeye simulations are unrealistically pessimistic, and/or that important lifecycle mechanisms such as density dependence or interactions with kokanee are not well modeled. Nonetheless, based on *PasRAS*

results—and common sense—the worse (and more stochastic) the marine and freshwater conditions are, and the lower the numbers and quality of initial seed stock used to establish the runs in the first place, the higher the juvenile collection efficacies would need to be to establish and maintain spring chinook and sockeye runs above Round Butte Dam.

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Jim Eisner, Bureau of Land Management

Mark Fritsch, Colleen Fagan, and Mike Gauvin, CTWS

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Amy Stuart, ODFW

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APPENDIX A. JUVENILE SOCKEYE SURVIVAL MODEL

APPENDIX A. SOCKEYE FRY SURVIVAL CALCULATIONS

The conceptual model for the interactions among sockeye and kokanee fry, bull trout, and zooplankton in the reservoir is shown in Figure A-1.

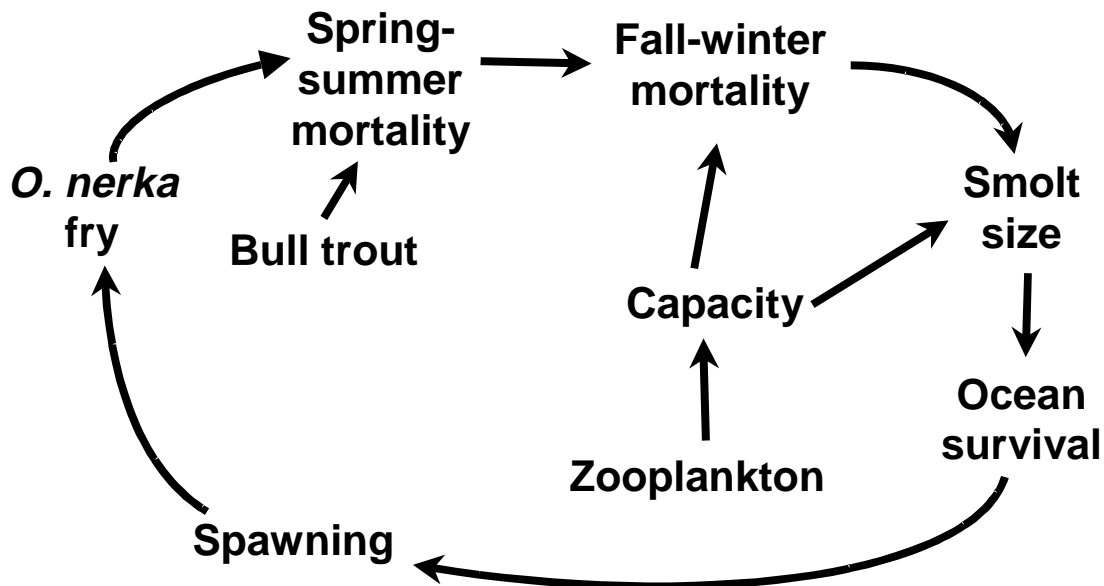


Figure A-1. Sockeye fry model.

Each simulated year, the computer algorithm proceeds as follows:

1. Calculate *O. nerka* Population

Estimate number of spring kokanee fry by sampling from a uniform probability distribution defined by minimum (*MinKokaneeFry*) and maximum (*MaxKokaneeFry*) parameter values defined by the user, and add it to the current population of sockeye spring fry:

$$KokaneeFry = \text{Uniform}(\text{MinKokaneeFry}, \text{MaxKokaneeFry})$$

$$NerkaPopulation = \text{SockeyeFry} + \text{KokaneeFry}$$

2. Calculate Spring Bull Trout Predation Mortality

Estimate constant predator take by sampling Z from a normal distribution defined by the average $PredationMean$ and coefficient of variation $PredationCV$ defined by the user:

$$Z = N(PredationMean, PredationMean * PredationCV)$$

If predation mortality Z is greater than the maximum fraction allowed $MaxPredationFraction$, then set predation to the maximum fraction allowed. Otherwise, set predation mortality M to the constant take Z :

If $Z/NerkaPopulation > MaxPredationFraction$

then

$$M = MaxPredationFraction$$

Else

$$M = Z$$

Implement mortality by sampling from a binomial distribution with $\hat{p} = M$.

3. Calculate minimum and maximum *O. nerka* populations.

Calculate the maximum *O. nerka* population possible, by dividing the maximum reservoir carrying capacity in kilograms $MaxResK$, by the minimum size that fry could be $MinFryLength$ times the (constant) mass assumed per mm of fry $FryMassPerLength$:

$$MaxPop = MaxResK / (MinFryLength * FryMassPerLength)$$

Similarly, calculate the minimum *O. nerka* population possible *MinPop*:

$$\text{MinPop} = \text{MinResK} / (\text{MaxFryLength} * \text{FryMassPerLength})$$

4. Calculate length of fall fry

Assume a linear relationship:

$$\text{FryLength} = m * \text{NerkaPopulation} + b$$

Where:

$$m = -(\text{MaxFryLength} - \text{MinFryLength}) / (\text{MaxPop} - \text{MinPop})$$

MaxFryLength = maximum length fall fry can be

MinFryLength - minimum length fall fry can be

MaxPop = maximum *O. nerka* fall fry population possible

MaxPop - *MinPop* = minimum *O. nerka* fall fry population possible

b = constant

An example curve for fry length as a function of population density is shown in Figure A-2.

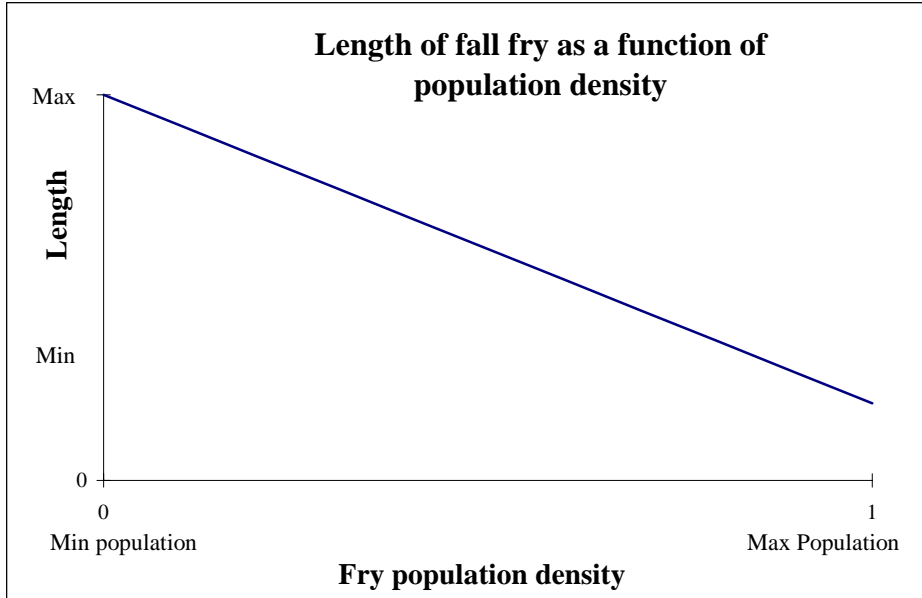


Figure A-2. Fry length as a function of population density (example curve).

5. Calculate *O. nerka* biomass

Multiply the number of fry *NerkaPopulation* times the mass per length *FryMassPerLength* times the fry length calculated in the previous step *FryLength*:

$$BioMass = NerkaPopulation * FryMassPerLength * FryLength$$

Add stochasticity by sampling from a normal distribution with a mean of *BioMass* and a C.V. of 0.15.

6. Calculate reservoir carrying capacity

Assume reservoir carrying capacity $K_{fry}(t)$ varies randomly between the minimum carrying capacity *MinResCarryingCapacity* and maximum carrying capacity *MaxResCarryingCapacity*, both defined by the user.

$$K_{fry}(t) = \text{Uniform}(\text{MinResCarryingCapacity}, \text{MaxResCarryingCapacity})$$

Where :

$K_{fry}(t)$ = carrying capacity of reservoir for *O. nerka* fall fry at time t , in kilograms

$MaxResCarryingCapacity$ = maximum reservoir carrying capacity, kg of *O. nerka*

$MinResCarryingCapacity$ = minimum reservoir carrying capacity, kg of *O. nerka*

7. Calculate density-dependent over-winter fry survival

Assume that fractional survival S_{fry} is a power function of *O. nerka* fall fry density:

$$S_{fry} = (1 - MaxMort) * X^r$$

Where:

$MaxMort$ = maximum mortality, when fry density is ≥ 1.0

r = initial slope

$$X = BioMass / K_{fry}(t)$$

The X axis value—i.e., the *O. nerka* fall fry density—is the fraction of the actual biomass $BioMass$ divided by the carrying capacity, $K_{fry}(t)$

An example curve for density-dependent over-winter survival is shown in Figure A-3.

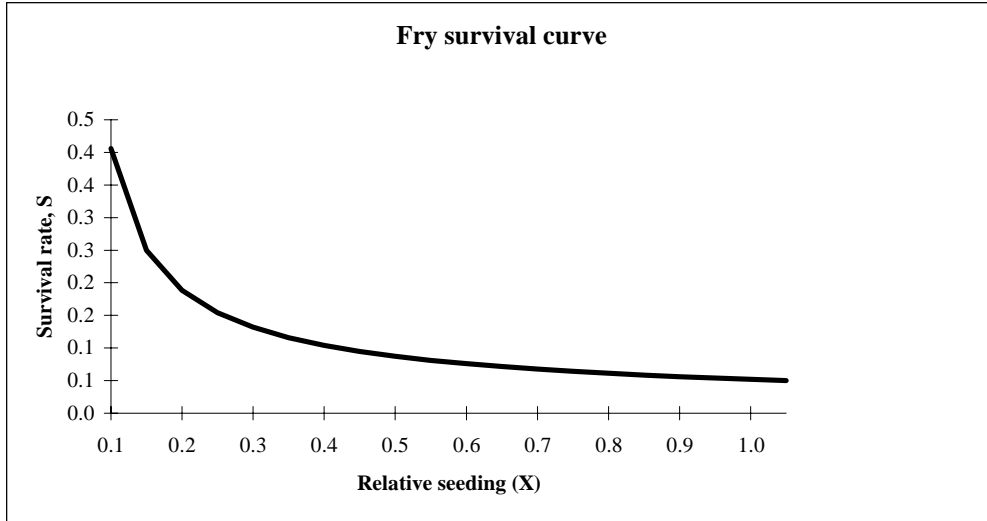


Figure A. 3. Density-dependent over-winter fry survival example curve.

APPENDIX B. SCENARIO PARAMETER SETTINGS

CHINOOK

Lower Granite chinook run reconstruction settings

Parameter name	Non-habitat dependent parameters	Good habitat	Fair habitat	Poor habitat	source
Smolt capacity	10,000,000				Schreck 1989
Initial spawners	40,000				Streamnet 1998
Fraction that stray away	0.1				
Fraction female spawners, high populations	0.5				
Fraction female spawners, small populations	0.5				Schreck 1989
Maximum fitness reduction for low population sizes	0.015				Nickelson 1998
Prespawning mortality	0.15				Schreck 1989
Nominal fecundity, mean	4500 (250)				Schreck 1989
Size of adults returning after 1 year (fecundity)	47.5 (1,861)				Mullan 1987
Size of adults returning after 2 years	69 (4,286)				Mullan 1987
Size of adults returning after 3 years	82 (6,302)				Mullan 1987
Egg-to-emergence mortality			0.2		Schreck 1989
Flood years per 100 years	0				
Juvenile catastrophic disease losses	0				
Irrigation diversion loss fraction (juveniles)	0				
Predation losses in tributaries (juveniles)	0				
Fraction that rear in reservoir	0				
Parr survival curves		Rearing habitat quality = "good"	Rearing habitat quality = "fair"	Rearing habitat quality = "poor"	
Constant (minimum survival)			0.016		Schreck 1989
Slope			-0.53		Schreck 1989
Minimum mortality			0.6		Schreck 1989
Parr survival CV	0.1				
Fraction of migrants attracted to forebay	1				
Fraction of migrants entrained	0				
Fraction of migrants collected	1				
Fraction of migrants successfully transported (Net collection efficacy)	1				Schreck 1989
Fraction of emmigrants lost in Deschutes	0				Schreck 1989
Fraction of emmigrants lost in Columbia	0.5				
Smolt catastrophic disease losses	0				
Estuary losses due to stress and disease	0				
Estuary losses due to predation (smoltification)	0				Derived from Mullan 1987
Fraction maturing in 1 year	0.037				Derived from Mullan 1987
Fraction maturing in 2 years	0.773				Derived from Mullan 1987
Fraction maturing in 3 years	0.19				
Adjust return times according to actual mortality?	No				Calibration
First year mortality	0.82				Calibration
Second year mortality	0.64				Calibration
Third year mortality	0.4				Calibration
ALPI adjustment: best year	0.78				Calibration
ALPI adjustment: 2nd best year	0.82				Calibration
ALPI adjustment: next to worst	0.85				Calibration
ALPI adjustment: worst	0.88				Schreck 1989
Harvest	0.15				
Adult catastrophic disease losses	0				Schreck 1989
Adult losses in the Columbia	0.4				Schreck 1989
Adult losses in the Snake	N/A				
Upstream transportation losses	0				

CHINOOK
Sherar's Falls chinook run reconstruction settings

Parameter name	Non-habitat dependent parameters	Good habitat	Fair habitat	Poor habitat	source
Smolt capacity	666,852				Fagan 1998
Initial spawners	3,000				Calibration
Fraction that stray away	0.1				
High fraction female spawners	0.4				
Low fraction female spawners	0.6				Nickelson 1998
Maximum fitness reduction for low population sizes	0.015				Nickelson 1998
Prespawning mortality	0.15				FTS
Nominal fecundity, mean	4500 (250)				Schreck 1998
Size of adults returning after 1 year (fecundity)	47.5 (1,861)				Mullan 1987
Size of adults returning after 2 years	69 (4,286)				Mullan 1987
Size of adults returning after 3 years	82 (6,302)				Mullan 1987
Egg-to-emergence mortality			0.2		Schreck 1998
Flood years per 100 years	0				
Juvenile catastrophic disease losses	0				
Irrigation diversion loss fraction (juveniles)	0				
Predation losses in tributaries (juveniles)	0				
Fraction that rear in reservoir	0				
Parr survival curves		Rearing habitat quality = "good"	Rearing habitat quality = "fair"	Rearing habitat quality = "poor"	
Constant (minimum survival)			0.016		Schreck 1998
Slope			-0.53		Schreck 1998
Minimum mortality			0.6		Schreck 1989
Parr survival CV	0.1				
Fraction of migrants attracted to forebay	1				
Fraction of migrants entrained	0				
Fraction of migrants collected	1				
Fraction of migrants successfully transported (Net collection efficacy)	1				
Fraction of emmigrants lost in Deschutes	0.1				FTS
Fraction of emmigrants lost in Columbia	0.2				FTS
Smolt catastrophic disease losses	0				
Estuary losses due to stress and disease	0				
Estuary losses due to predation (smoltification)	0				
Fraction maturing in 1 year	0.017				Fagan 1998
Fraction maturing in 2 years	0.7049				Fagan 1998
Fraction maturing in 3 years	0.278				
Adjust return times according to actual mortality?	No				
First year mortality	0.89				Calibration
Second year mortality	0.64				Calibration
Third year mortality	0.4				Calibration
ALPI adjustment:best year	0.88				Calibration
ALPI adjustment:2nd best year	0.89				Calibration
ALPI adjustment: next to worst	0.92				Calibration
ALPI adjustment: worst	0.93				Calibration
Harvest	0.3				Fagan 1998
Adult catastrophic disease losses	0				
Adult losses in the Columbia	0.08				FTS
Adult losses in the Deschutes	0				
Upstream transportation losses	0				

CHINOOK

Scenario 0: FTS recommendations, based on Sherar's falls reconstruction settings, with Habrate database

Scenario 1: same as #0 except collection efficacy reduced to 0.6

Scenario 2: same as #1, except initializing with smaller populations

Scenario 3: same as #0, except with smolt capacity reduced by 20%.

Parameter name	Non-habitat dependent parameters	Good habitat	Fair habitat	Poor habitat	source
Smolt capacity	331,873				Riehle 1999
Initial spawners	33,187				Riehle 1999
Fraction that stray away	0.1				
High fraction female spawners	0.4				
Low fraction female spawners	0.6				Nickelson 1998
Maximum fitness reduction for low population sizes	0.015				Nickelson 1998
Prespawning mortality	0.15				FTS
Nominal fecundity, mean	4500 (250)				Schreck 1998
Size of adults returning after 1 year (fecundity)	47.5 (1,861)				Mullan 1987
Size of adults returning after 2 years	69 (4,286)				Mullan 1987
Size of adults returning after 3 years	82 (6,302)				Mullan 1987
Egg-to-emergence mortality		0.15	0.5	0.85	FTS
Egg-to-emergence mortality, flood years					
Flood years per 100 years	0				
Juvenile catastrophic disease losses	0				
Irrigation diversion loss fraction (juveniles)	0				
Predation losses in tributaries (juveniles)	0				
Fraction that rear in reservoir	0				
Parr survival curves		Rearing habitat quality = "good"	Rearing habitat quality = "fair"	Rearing habitat quality = "poor"	
Constant (minimum survival)		0.025	0.016	0.02	FTS
Slope		-0.6	-0.53	0	FTS
Minimum mortality		0.6	0.6	0.6	FTS
Parr survival CV	0.1				
Fraction of migrants attracted to forebay	1				
Fraction of migrants entrained	0				
Fraction of migrants collected	1				
Fraction of migrants successfully transported	1				
(Net collection efficacy)	1				
Fraction of emmigrants lost in Deschutes	0.1				FTS
Fraction of emmigrants lost in Columbia	0.2				FTS
Smolt catastrophic disease losses	0				
Estuary losses due to stress and disease	0				
Estuary losses due to predation (smoltification)	0				
Fraction maturing in 1 year	0.017				Fagan 1998
Fraction maturing in 2 years	0.7049				Fagan 1998
Fraction maturing in 3 years	0.278				
Adjust return times according to actual mortality?	No				
First year mortality (CV)	0.89 (0.01)				Calibration
Second year mortality	0.64 (0.1)				Calibration
Third year mortality	0.4 (0.1)				Calibration
ALPI adjustment:best year	0.88 (0.1)				Calibration
ALPI adjustment:2nd best year	0.89 (0.1)				Calibration
ALPI adjustment: next to worst	0.92 (0.1)				Calibration
ALPI adjustment: worst	0.93 (0.1)				Calibration
Harvest	0.3				Fagan 1998
Adult catastrophic disease losses	0				
Adult losses in the Columbia	0.08				FTS
Adult losses in the Deschutes	0				
Upstream transportation losses	0				

SOCKEYE

Kenai River sockeye run reconstruction settings

Parameter name	Non-habitat dependent parameters	Good habitat	Fair habitat	Poor habitat	Source
Fraction that stray away		0.05	0.05	0.05	
High fraction female spawners	0.50				Mundy 1998
Low fraction female spawners	0.50				
Maximum fitness reduction for low population sizes					Nicholson 1998
Prespawning mortality	0.05				Mundy 1998
Nominal fecundity, mean	3150 (472.5)				Mundy 1998
Fraction of spawning habitat assumed to be good, fair or poor		0.8	0.2	0	Mundy 1998
Egg-to-emergence mortality		0.5	0.5	n/a	Mundy 1998
Emergence-to-onset-of-feeding mortality		0	0	n/a	
Juvenile catastrophic disease losses	0.00				
Irrigation diversion loss fraction (juveniles)	0.00				
Predation, tribs	0.65				Mundy 1998
Summer: constant predation	320,000,000				Mundy 1998-1999: 10% of capacity
Summer: maximum predation fraction	0.80				Mundy 1998-1999
Fry mass/length, g/mm	0.04				Mundy 1998-1999
Min fry length, mm	49.00				Mundy 1998-1999
Max fry length, mm	89.00				Mundy 1998-1999
Min reservoir biomass capacity, kg fall 0	78,400				10% of max
Max reservoir biomass capacity, kg fall 0	784,000				400M fry (Mundy)* smallest size
Min kokanee population	0.00				
Max kokanee population	0.00				
Overwinter constant	0.10				Derived from Foerster 1968, Burgner 1991
Overwinter slope	-0.20				Derived from Foerster 1968, Burgner 1991
Fraction of migrants attracted to forebay	1.00				
Fraction of migrants entrained	0.00				
Fraction of migrants collected	1.00				
Fraction of migrants successfully transported (Net collection efficacy)	1.00				
Fraction of emmigrants lost in Deschutes	0.00				Mundy 1998
Fraction of emmigrants lost in Columbia	0.00				Mundy 1998
Smolt catastrophic disease losses	0.00				
Estuary losses due to stress and disease	0.00				
Estuary losses due to predation (smoltification)	0.00				
Marine exploitation rate, annual	0.20				Burgner 1991
Fraction maturing in 1 year	0.02				Burgner 1991
Fraction maturing in 2 years	0.23				Burgner 1991
Fraction maturing in 3 years	0.76				Burgner 1991
Size of adults returning after 1 year	56.00				Burgner 1991
Size of adults returning after 2 years	58.78				Burgner 1991
Size of adults returning after 3 years	61.56				Burgner 1991
Fecundity of adults returning after 1 year	2908.8 (436.32)				Mundy 1998-1999
Fecundity of adults returning after 2 years	3150.104 (472.5)				Mundy 1998-1999
Fecundity of adults returning after 3 years	3391.408 (508.7)				Mundy 1998-1999
First year mortality (big smolts, small smolts, CV)	(0.58, 0.7, 0.1)				Calibration
Second year mortality (big smolts, small smolts, CV)	(0.6, 0.68, 0.15)				Calibration
Third year mortality (big smolts, small smolts, CV)	(0.512, 0.61, 0.15)				Calibration
ALPI adjustment: best year	0.60				Calibration
ALPI adjustment: 2nd best year	0.70				Calibration
ALPI adjustment: next to worst	0.90				Calibration
ALPI adjustment: worst	0.95				Calibration
Initial escapement, best case	10,000,000.00				ADFG 1998
Initial escapement, median case	2,500,000.00				ADFG 1998
Initial escapement, worst case	200,000.00				ADFG 1998
Adult catastrophic disease losses	0.00				
Adult losses in the Columbia	0.05				Mundy 1998
Adult losses in the Deschutes	0.00				Mundy 1998
Upstream transportation losses	0.00				

SOCKEYE

Okanagon sockeye run reconstruction settings

Parameter name	Non-habitat dependent parameters	Good habitat	Fair habitat	Poor habitat	Source
Fraction that stray away		0.05	0.05	0.05	
High fraction female spawners	0.60				Mundy 1998
Low fraction female spawners	0.40				Mundy 1998
Maximum fitness reduction for low population sizes	0.02				Nicholson 1998
Prespawning mortality	0.15				Mundy 1998
Nominal fecundity, mean	2900 (424)				Fryer 1995
Fraction of spawning habitat assumed to be good, fair or poor		0.8	0.2	0	Ratliff 1998
Egg-to-emergence mortality		0.5	0.6	n/a	Derived from Fryer 1995
Emergence-to-onset-of-feeding mortality		0	0	n/a	
Juvenile catastrophic disease losses	0.00				
Irrigation diversion loss fraction (juveniles)	0.00				
Predation, tribes	0.30				Calibration
Summer: constant predation	1,900,000				Mundy 1998-1999
Summer: maximum predation fraction	0.90				Mundy 1998-1999
Fry mass/length, g/mm	0.14				Derived from Burgner 1991, assuming large fry
Min fry length, mm	80.00				Ratliff 1998
Max fry length, mm	120.00				Ratliff 1998
Min reservoir biomass capacity, kg fall 0	24,192				10% of max
Max reservoir biomass capacity, kg fall 0	241,920				21,600,000 fry (Fryer 1995) *min fry length
Min kokanee population	0.00				
Max kokanee population	0.00				
Overwinter constant	0.10				Derived from Foerster 1968, Burgner 1991
Overwinter slope	(0.20)				Calibration
Maximum mortality, overwinter	0.60				
Fraction of migrants attracted to forebay	1.00				
Fraction of migrants entrained	0.00				
Fraction of migrants collected	1.00				
Fraction of migrants successfully transported	1.00				
(Net collection efficacy)	1.00				
Fraction of emmigrants lost in Deschutes	0.00				
Fraction of emmigrants lost in Columbia	0.65				Mundy 1998
Smolt catastrophic disease losses	0.00				
Estuary losses due to stress and disease	0.00				
Estuary losses due to predation (smoltification)	0.00				
Marine exploitation rate, annual	0.05				Fryer 1995
Fraction maturing in 1 year	0.07				Fryer 1995
Fraction maturing in 2 years	0.89				Fryer 1995
Fraction maturing in 3 years	0.04				Fryer 1995
Size of adults returning after 1 year	46.69				Fryer 1995
Size of adults returning after 2 years	55.66				Fryer 1995
Size of adults returning after 3 years	64.07				Fryer 1995
Fecundity of adults returning after 1 year	2,014.00				Mundy 1998-1999
Fecundity of adults returning after 2 years	2,879.00				Mundy 1998-1999
Fecundity of adults returning after 3 years	3,609.00				Mundy 1998-1999
First year mortality (big smolts, small sm)	(0.6, 0.96, 0.1)				Calibration
Second year mortality (big smolts, small s)	(0.5, 0.65, 0.15)				Calibration
Third year mortality (big smolts, small sn)	(0.48, 0.6, 0.15)				Calibration
ALPI adjustment: best year	0.70				Calibration
ALPI adjustment: 2nd best year	0.80				Calibration
ALPI adjustment: next to worst	0.93				Calibration
ALPI adjustment: worst	0.96				Calibration
Initial escapement, best case	129,600.00				Streamnet 1998
Initial escapement, median case	34,852.00				Streamnet 1998
Initial escapement, worst case	1,665.00				Streamnet 1998
Adult catastrophic disease losses	0.00				
Adult losses in the Columbia	0.25				Mundy 1998-1999
Adult losses in the Deschutes	0.00				Mundy 1998-1999
Upstream transportation losses	0.00				

SOCKEYE

Lake Billy Chinook sockeye settings

Parameter name	Non-habitat dependent parameters	Good habitat	Fair habitat	Poor habitat	Source
Fraction that stray away		0.05	0.05	0.05	
High fraction female spawners	0.60				Sockeye work group
Low fraction female spawners	0.40				Sockeye work group
Maximum fitness reduction for low population sizes	0.02				Nicholson 1998
Prespawning mortality	0.15				Sockeye work group
Nominal fecundity, mean	2900 (424)				Fryer 1995
Fraction of spawning habitat assumed to be good, fair or poor		0.8	0.2	0	Sockeye work group
Egg-to-emergence mortality		0	0	n/a	Subsumed in "Predation, tribs"
Emergence-to-onset-of-feeding mortality		0	0	n/a	
Juvenile catastrophic disease losses	0.00				
Irrigation diversion loss fraction (juveniles)	0.00				
		Total egg-to-reservoir mortality			
Predation, tribs	0.95				Kern 1999 Subsumed in "Predation, tribs"
Summer: constant predation	0 (CV=.15)				Sockeye work group
Summer: maximum predation fraction	0.30				Kern 1999
Fry mass/length, g/mm	0.11				Kern 1999
Min fry length, mm	76.00				Thiesfeld 1999
Max fry length, mm	171.00				10% of max
Min reservoir biomass capacity, kg fall 0	2,044				Thiesfeld 1999
Max reservoir biomass capacity, kg fall 0	20,439				
Min kokanee population	0.00				Based on Thiesfeld 1999
Max kokanee population	1,824,920				Kern 1999
Overwinter constant	0.32				Sockeye work group
Overwinter slope	-0.20				Kern 1999
Maximum mortality, overwinter	0.60				
Fraction of migrants attracted to forebay	1.00				
Fraction of migrants entrained	0.00				
Fraction of migrants collected	1.00				
Fraction of migrants successfully transported (Net collection efficacy)	1.00				
Fraction of emmigrants lost in Deschutes	0.08				Mundy 1998-1999
Fraction of emmigrants lost in Columbia	0.28				Mundy 1998-1999
Smolt catastrophic disease losses	0.00				
Estuary losses due to stress and disease	0.00				
Estuary losses due to predation (smoltification)	0.00				
Marine exploitation rate, annual	0.00				Fryer 1995
Fraction maturing in 1 year	0.07				Fryer 1995
Fraction maturing in 2 years	0.89				Fryer 1995
Fraction maturing in 3 years	0.04				Fryer 1995
Size of adults returning after 1 year	46.69				Fryer 1995
Size of adults returning after 2 years	55.66				Fryer 1995
Size of adults returning after 3 years	64.07				Fryer 1995
Fecundity of adults returning after 1 year	2,014.00				Mundy 1998-1999
Fecundity of adults returning after 2 years	2,879.00				Mundy 1998-1999
Fecundity of adults returning after 3 years	3,609.00				Mundy 1998-1999
First year mortality (big smolts, small smolts, CV)	(0.6, 0.96, 0.1)				Calibration
Second year mortality (big smolts, small smolts, CV)	(0.5, 0.65, 0.15)				Calibration
Third year mortality (big smolts, small smolts, CV)	(0.48, 0.6, 0.15)				Calibration
ALPI adjustment:best year	0.70				Calibration
ALPI adjustment:2nd best year	0.80				Calibration
ALPI adjustment: next to worst	0.93				Calibration
ALPI adjustment: worst	0.96				Calibration
Initial escapement, best case	56,000.00				Thiesfeld 1999
Initial escapement, median case	22,000.00				Thiesfeld 1999
Initial escapement, worst case	17,000.00				Thiesfeld 1999
Adult catastrophic disease losses	0.00				
Adult losses in the Columbia	0.08				Mundy 1998-1999
Adult losses in the Deschutes	0.01				Mundy 1998-1999
Upstream transportation losses	0.00				

APPENDIX C. *HABRATE* SETTINGS FOR SPRING CHINOOK

Appendix C. Habrate as modified by Mike Riehle

Rating	Habitat Quality	Chinook Survival		
		Spawning to Emergence	0+ Summer	0+ Winter
3	Good	0.85	0.80	0.50
2	Fair	0.50	0.50	0.25
1	Poor	0.15	0.20	0.10

0 Not available

Stream-Reach Number	HABITAT QUALITY		STRAYING	SMOLT	Initial spawners
	Egg/Fry	Fry/Parr	System	CAPACITY	
CROOKED RIVER1	2	2	3	4554.1656	455.41656
CROOKED RIVER2	2	3	3	34214.8782	3421.48782
CROOKED RIVER3	2	2	3	29771.925	2977.1925
CROOKED RIVER4	2	2	3	12842.8125	1284.28125
CROOKED RIVER5	2	1	3	4490.037	449.0037
CROOKED RIVER6	2	1	3	7459.074	745.9074
CROOKED RIVERUNS-7	1	1	3	61869.85	6186.985
CROOKED RIVER8	2	1	3	2399.658	239.9658
CROOKED RIVER9	2	1	3	6.06905	0.606905
CROOKED RIVER10	2	1	3	6237.6678	623.76678
MCKAY CREEK1	1	1	3	2372.7896	237.27896
MCKAY CREEKUNS-2	1	2	3	663.75	66.375
MCKAY CREEK3	1	1	3	2.8575	0.28575
MCKAY CREEKUNS-4	1	1	3	847.3815	84.73815
MCKAY CREEK5	1	1	3	260.27055	26.027055
MCKAY CREEKUNS-6	1	2	3	7397.376	739.7376
MCKAY CREEK7	3	2	3	2472.597	247.2597
DESCHUTES RIVER1	2	1	2	8491.728	849.1728
DESCHUTES RIVER2	2	2	2	30166.0965	3016.60965
DESCHUTES RIVER3	2	1	2	0	0
SQUAW CREEK1	2	1	2	2427.612	242.7612
SQUAW CREEK2	2	1	2	1915.689	191.5689
SQUAW CREEK3	2	1	2	1948.767	194.8767
SQUAW CREEKUNS - 4	1	1	2	4744.74	474.474
SQUAW CREEK5	2	1	2	2095.64775	209.564775
SQUAW CREEKUNS - 6	2	1	2	886.5792	88.65792
SQUAW CREEK7	2	1	2	368.373	36.8373
SQUAW CREEKUNS - 8	2	1	2	356.44	35.644
SQUAW CREEK9	2	1	2	476.14	47.614
SQUAW CREEKUNS - 10	2	2	2	425.536	42.5536
SQUAW CREEK11	3	1	2	1504.0305	150.40305
SQUAW CREEK12	3	1	2	4247.474	424.7474
ABBOTT CREEK1	2	1	1	965.2095	96.52095
ABBOTT CREEK2	1	1	1	0	0
ABBOTT CREEK1	2	1	1	965.2095	96.52095
ABBOTT CREEK2	1	1	1	0	0
BEAR VALLEY CREEK1	1	1	1	0	0
BRUSH CREEK1	1	1	1	478.818	47.8818
BRUSH CREEKDRY-2	1	0	1	0	0
BRUSH CREEK3	1	1	1	0	0
BRUSH CREEK4	1	1	1	0	0
CABOT CREEK1	1	1	1	0	0
CANDLE CREEK1	2	1	1	693.501	69.3501
CANDLE CREEK2	2	1	1	2549.3325	254.93325
CANDLE CREEK3	2	2	1	603.879	60.3879
CANDLE CREEK4	1	2	1	0	0
CANDLE CREEK5	1	1	1	0	0
CANYON CREEK1	3	2	1	5534.77	553.477
CANYON CREEK2	3	2	1	1620.987	162.0987
CANYON CREEK3	1	1	1	0	0
CANYON CREEK4	1	2	1	2955.5875	295.55875
CANYON CREEK5	1	1	1	0	0
FIRST CREEKDRY-1	2	1	1	152.5	15.25

FIRST CREEK2	3	1	1	5163.3065	516.33065
FIRST CREEK3	1	1	1	0	0
INDIAN FORD CREEK DRY-1	1	0	2	0	0
INDIAN FORD CREEK UNRS-2	1	0	2	176.484	17.6484
INDIAN FORD CREEK3	1	2	2	2099.388	209.9388
JACK CREEK1	1	1	1	3614.112	361.4112
JACK CREEK 19892	1	1	1	353.8865	35.38865
JEFFERSON CREEK1	2	1	1	1393.536	139.3536
JEFFERSON CREEK2	2	1	1	713.115	71.3115
JEFFERSON CREEK3	2	1	1	1469.637	146.9637
JEFFERSON CREEK4	1	1	1	320.183	32.0183
JEFFERSON CREEK5	1	1	1	0	0
LAKE CREEK - NORTH FORK UNRS-2	1	0	1	3907.2	390.72
LAKE CREEK - MIDDLE FORK UNRS-3	1	0	1	4673.2	467.32
LAKE CREEK - SOUTH FORK UNRS-1	1	0	1	772.8	77.28
LAKE CREEK - SOUTH FORK5	2	1	1	1482.6525	148.26525
LAKE CREEK - NORTH FORK6	1	1	1	881.894	88.1894
LAKE CREEK - MIDDLE FORK7	1	1	1	151.422	15.1422
LAKE CREEK8	1	1	1	269.742	26.9742
LINK CREEK1	2	2	1	1449.947	144.9947
METOLIUS RIVER11	2	1	1	26547.2595	2654.72595
METOLIUS RIVER12	2	1	1	3758.02	375.802
METOLIUS RIVER UNRS-13	2	0	1	196.812	19.6812
METOLIUS RIVER14	2	1	1	1479.0875	147.90875
POLE CREEK1	1	1	2	677.908	67.7908
POLE CREEK2	1	1	2	142.362	14.2362
ROARING CREEK1	1	1	1	307.725	30.7725
ROARING CREEK2	1	1	1	20.7095	2.07095
SNOW CREEK1	1	1	2	0	0
SNOW CREEK2	1	1	2	0	0
SNOW CREEK3	1	1	2	0	0
SQUAW CREEK13	2	1	2	701.7192	70.17192
SQUAW CREEK14	2	1	2	2365.77	236.577
SQUAW CREEK15	2	1	2	687.361	68.7361
SQUAW CREEK16	2	1	2	544.467	54.4467
SQUAW CREEK17	2	1	2	458.88	45.888
MCKAY CREEK8	2	1	3	1211.6325	121.16325
MCKAY CREEK UNRS-9	1	0	3	119.83832	11.983832
MCKAY CREEK10	1	1	3	550.812	55.0812
MCKAY CREEK11	1	1	3	251.5125	25.15125
MCKAY CREEK12	1	1	3	155.324	15.5324
MCKAY CREEK13	1	1	3	468.318	46.8318
LITTLE MCKAY CREEK1	2	1	3	347.652	34.7652
LITTLE MCKAY CREEK2	1	1	3	405.2525	40.52525
LITTLE MCKAY CREEK3	2	1	3	213.528	21.3528
LITTLE MCKAY CREEK4	2	1	3	312.243	31.2243
LITTLE MCKAY CREEK5	1	1	3	0	0
LITTLE MCKAY CREEK6	1	1	3	0	0
HEISING SPRING1	1	1	1	0	0
SPRING CREEK1	2	1	1	296.415	29.6415
FLY CREEK UNRS-1	0	0	1	0	0
FLY CREEK DRY-2	0	0	1	0	0
FLY CREEK DRY-3	0	0	1	0	0
FLY CREEK4	0	0	1	0	0
WHITEWATER RIVER	1	1	1	3609.34	360.934
WHITEWATER RIVER	1	1	1	1644.3195	164.43195
WHITEWATER RIVER	1	1	1	1068.095	106.8095