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Maximum Sustained Yield or Biologically Based Escapement Goals for Selected Chinook Salmon Stocks Used by the Pacific Salmon Commission's Chinook Technical Committee for Escapement Assessment

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INTRODUCTION

In February of 1998, the United States (U.S.) and Canada exchanged proposals regarding management regimes for chinook salmon. The similarities and differences of the two proposals were discussed in TCCHINOOK (98)-01. In addition to exchanging proposed management regimes, both parties also instructed the Chinook Technical Committee (CTC) to determine Maximum Sustained Yield (MSY) or other biologically based escapement goals for the 46 chinook stocks that the CTC uses to assess fisheries effects upon wild chinook salmon escapements. In TCCHINOOK (98)-01, the CTC identified eight stocks (Situk, Alsek, King Salmon, Unuk, Chickamin, Keta, Blossom, and Andrew Creek) for which there already existed agreed MSY escapement goals.

This report includes a chapter explaining the general methods for stock-recruitment analysis and the resulting MSY or biologically based escapement goals for seven additional escapement assessment stocks: Taku, Stikine, Lewis, Columbia River Summer, Nehalem, Siletz, and Siuslaw.

1. GENERAL METHODS FOR STOCK-RECRUIT ANALYSIS

1.1 Introduction

"What is the spawning abundance that produces maximum sustained yield from a salmon population?" is a scientific question that has traditionally been treated as a statistical problem. Data on production and spawning abundance are analyzed through regression models to estimate parameters of a mathematical model that describes the relationship between spawning abundance and production. The first derivative of the mathematical model is solved for the spawning abundance that produces maximum surplus production (S_{MSY}).

Although the basic description of statistical stock-recruit analysis can be simply stated, arriving at unbiased estimates from such analysis is far from simple. Production and spawning abundance can be estimated in various ways or expressed in different terms. Further, production from salmon is often influenced greatly by factors other than spawning escapement and observations on spawning escapements can be confined to a relatively narrow range due to management actions that are intended to achieve target goals. Variables are often measured with error and are sometimes auto-correlated. If undetected or ignored, these problems can result in seriously biased estimates. Hilborn and Walters (1992:section 7.8) provide a very readable discussion on most of these problems. The key to a successful stock-recruit analysis is to develop unbiased estimates of variables, and when estimates are biased, to find a means of accommodating these biases.

Described below are some simple steps for statistically analyzing stock-recruit data to estimate S_{MSY}. These steps encapsulate some of the thinking used in the statistical analyses for stocks in the following chapters. Quinn and Deriso (1999:Chapter 3) is an excellent reference for the mathematical and statistical methods described below. Other "thinking" is of course possible, and in fact should be encouraged, so long as it leads to unbiased, defensible estimates in a timely manner. For instance, guidelines and advice given below are predicated on the standard frequentist approach to regression analysis, however, some of the same steps are relevant to a Bayesian analysis. Frequentist and Bayesian regression analyses should provide essentially the same estimates, however, the latter type of analysis is not as developed as the former for stock-recruit type data. Markov tables and "fuzzy" logic are other approaches that have been recently adapted to analyze stock-recruit data.

The intent of this discourse is not to provide a limited set of instructions, but to provide a general introduction to our efforts to determine biologically based escapement goals for stocks of chinook salmon. Hopefully, the steps below will help us recognize what information is essential for our analyses, when we have that information, and what can be done to correct deficiencies.

1.2 Estimating Spawning Abundance

Like most species of Pacific salmon, chinook salmon are semelparous, that is they die after spawning. Abundance over the spawning grounds may be defined as mature chinook salmon, mature females, "large" chinook salmon, or as eggs produced. Often the number of precocious males (jacks) are not estimated or intentionally ignored in estimating parameters for stock-recruitment relationships. Males in general may be considered "surplus" as spawners, so long as there are enough to service the females present. Eggs produced are an intuitively appealing measure of spawning abundance as this is the first stage in the life history of a new year class. The appropriate measure for spawning abundance is often a subjective choice, based on convenience and the availability of data, depending upon the purpose of the stock-recruitment analysis.

Annual stock assessments to estimate abundance of spawning chinook salmon is based on counting or sampling mature chinook salmon or indexing their abundance:

Counting all chinook salmon is usually associated with some sort of barrier to their upstream spawning migration: a weir, a dam, or an entry chute to a raceway if the population is from a hatchery. Unimpeded passage can be counted with sonar or from towers or bridges.

Sampling to estimate abundance can be performed at a migration barrier, but can also be associated with a mark-recapture experiment or spawning escapement survey. Not all salmon passing a barrier need be counted. If all passing salmon are counted only during systematically scheduled periods throughout a day or week, then the average count can be expanded upward to account for times with no sampling. In mark-recapture experiments, a sample of chinook salmon are marked and released back into the population. The fraction of marked individuals in a later sample is then multiplied by the number marked to estimate abundance.

Indexing is a partial count over the spawning grounds from the air, on foot, or while floating downstream in a boat or in a wet suit wearing snorkeling gear. Spawning or spawned-out chinook salmon or their redds are counted. All or part of the spawning grounds are surveyed one to a few times to produce the index, usually calculated as the average or maximum (peak) of several counts. Because such an index is an underestimate of spawning abundance, some independent information gathered beyond the survey is needed to expand the index, usually at the cost of some added imprecision in estimates.

When the same area is surveyed repeatedly during a spawning season, an area-under-the curve method can be employed to expand the index to estimate escapement. Partial counts are summed (integrated) across surveys, then the sum adjusted for the length of time the thing being counted is expected to have existed. For partial counts of salmon, an estimate of stream life is used to adjust sums; for redd counts, it's an estimate of redd life.

An index can also be expanded through calibration. Peak, average, or summed partial counts are compared against estimates of abundance from mark-recapture experiments or from counts at weirs or dams to produce an average expansion factor for the index. This factor is then used for years with no mark-recapture experiments or no counts at dams or weirs.

Spawning escapements are usually not known with certainty, but rather have an associated uncertainty (expressed as a statistical variance) that may or may not be quantified. This variance is one type of *measurement error* (see Section 1.5).

Often the estimated escapement must be partitioned into several groups, and sampling is again used to split the abundance estimate. A representative sample is drawn from the frame (all the fish that could be sampled), and an attribute is measured on each salmon that distinguishes its membership in a group; for example, a sex, an age, or a stock. Commonly used attributes to partition escapements include gender, skin coloration, morphological and meristic characters, scale patterns, otolith patterns, parasite associations, CWT recoveries, or genetic markers.

The fraction of the sample represented by a group is multiplied by the estimate of total escapement to produce an estimate of escapement for that group. With chinook salmon, escapements should be partitioned into escapements by age to obtain the data necessary to perform S-R analysis. For instance, if spawning abundance is defined as females only, an estimate of the number of redds is an appropriate measure of spawning abundance. However, if spawning abundance is defined as females plus males (as is often so), the estimated number of redds must be divided by an estimate of the fraction of the population comprised of females to estimate spawning abundance. The fraction of the sample holding the same attribute is itself an estimate with its own sampling variance which adds to the overall measurement error. Some useful equations on calculating these estimates are given in Section 1.5.

Sampling mature chinook salmon is also subject to another kind of measurement error: bias. Statistical methods for observational studies are based on a randomly drawn sample, however, commonly available data are not produced by random sampling. Sampling chinook salmon is usually opportunistic, that is, some gear is used to capture fish over some location at some time. Sampling gears such as gillnets, fishwheels, and carcass weirs are size and sex selective. Sampling at certain times and places can also produce bias, as is often the case with carcass surveys and sampling over only part of the spawning grounds.

Even though sampling is not random, there are ways bias can be avoided if sampling is representative. Some common schemes used to promote representative sampling or correct bias estimates are:

- Systematically sampling over the entire frame (population)
  - Sampling every kth fish passing through a weir/dam or caught in fishwheels
  - Counting fish in every kth section of spawning grounds

- Stratified systematic sampling over time or space
  - Sampling a systematically drawn subset of all fish passing through a weir/dam or caught in a fishwheel every kth day (a 24 hr day)
  - Sampling all fish caught in gillnets or seines every kth day with equal fishing effort each day

- Sampling in mark-recapture experiments to adjust size/sex/age selective sampling in:
  - Carcass surveys
  - Carcass weirs
  - Gillnets
  - Seines
  - Hook and Line
  - Fishwheels

- Radio telemetry to find extent of spawning grounds

These schemes are often used in combination to correct bias due to sampling opportunistically. In systematic sampling, sampling units are selected regardless of how difficult the logistics of sampling may be, short of being impossible. Sampling with equal sampling effort means equal amounts of soak time for nets and lines. When nets or lines are being retrieved to process fish, they are not being "soaked." Recapture of marked fish can be used to estimate probabilities of capture which in turn can be used to "correct" information from size or sex-selective sampling.

1.3 Estimating Production

While spawning abundance may have any of a number of definitions, production is measured in fish caught and killed, and fish surviving to spawn. Production by a year class is the sum of the following:

- Reported or estimated landed catch in fisheries
- Salmon caught in fisheries, released, then subsequently dying from the experience
- Salmon that die from contact with the gear, but are not brought to the vessel (e.g., predator loss, net drop out)
- Salmon that survive to spawn

In terminal fisheries on mature chinook salmon, reported commercial harvest is a tally while harvest in recreational fisheries is usually estimated, and so has a sampling variance. Unreported mortalities due to fishing are usually estimated with a factor tied to fishing effort or reported harvest. These factors are estimates from independent studies and also have a sampling variance.

All of the above considerations hold for mixed-stock fisheries as well, plus there must be some means of segregating harvests by stock, age, and maturity. Genetic patterns can be used to estimate the fraction of a harvest comprised of a stock or group of stocks. Since this fraction is estimated from a sample of the harvest, sampling variance is involved. The population of interest can be tagged directly to produce estimates of harvest. Alternatively, cohort analysis, based on coded-wire-tag (CWT) recoveries of an associated "indicator" stock can be used to generate estimates of production of natural runs through cohort reconstruction techniques. Usually, the "indicator" stock is a hatchery stock that is presumed to be representative of the population of interest – that is the stocks suffer the same exploitation pattern, the same marine survival rates, and the same maturation schedule. Production is estimated by reconstruction of the run by dividing the terminal run by age by the complement of the exploitation rate (the survival rate) (Appendix A.1). Estimates of production are then converted into adult equivalents using estimated maturation and survival rates obtained from cohort analysis.

Production from a given level of spawning escapement can vary substantially due to environmental influences. Changes in the environment over time can impact trends in productivity and carrying capacity. If production varies randomly about a central tendency, then variability is a form of *process error*. If survival estimates for the stock or a representative surrogate are available, survival rates can be included in the stock-recruit analysis to better reveal the relationship between production and spawning abundance (see Section 1.6).

1.4 Investigating Contrast

There must be sufficient contrast in the data for stock-recruit analysis to have any reasonable hope of estimating S_{MSY}. Any stock-recruit analysis is pointless unless there has been meaningful variation in spawning abundance. Here the term "meaningful" is relative. Estimated spawning abundance in the Stikine River has ranged just over an order of magnitude from 5,700 to 57,000 large spawners while estimated production ranged just under an order of magnitude from 7,500 to 72,000. Spawning abundance in the Stikine River largely reflects the natural range of production, as would be expected of a population exposed to limited exploitation. Not so for the early years in the data series for the population spawning in the Hanford Reach of the Columbia River. From 1968 through 1984 spawning abundance ranged from 14,000 to 48,000 while production went from 56,000 to 957,000. Such a high ratio of production to spawning abundance indicates that data from the Hanford Reach prior to 1985 represents only a limited segment of the underlying stock-recruit relationship. In general, the following guidelines may be useful:

- When estimates of spawning abundance are similar – range is less than 4 times the smallest spawning abundance – statistical stock-recruit analysis is likely to produce a poor estimate of S_{MSY}.
- When range in spawning abundance is 4 to 8 times the smallest level, statistical stock-recruit analysis should produce better estimates of S_{MSY}, so long as measurement error is not extreme and some of the production-to-spawner ratios are below one at higher levels of spawning abundance.
- When range is >8, statistical analysis should produce the best estimates, so long as some of the production-to-spawner ratios are below one at higher levels of spawning abundance.

Another way to evaluate contrast is to use ratios of production per spawner as a means of estimating S_{MSY}. Production-to-spawner ratios below "replacement" are circumstantial, but not conclusive evidence of density-dependent survival rates. Likewise, lack of these low ratios is circumstantial evidence that spawning abundance has not been high enough to expose the underlying density-dependent relationship.

If contrast is small, the estimate of S_{MSY} will be determined by process error, an extreme environmental event, or by a bad case of measurement error, not by the underlying relationship between spawning abundance and production. In this case, the estimate of S_{MSY} will be unreliable and may be seriously biased.

Consistently high production-to-spawner ratios for an exploited population indicate that spawning abundance has been constrained to low levels and that there will be insufficient contrast to accurately estimate the parameters of a stock-recruit relationship. However, a few production-to-spawner ratios less than one do not imply that the data have sufficient contrast. Density-independent factors might have driven production below replacement, even when spawning abundance was low.

1.5 Investigating Measurement Error

Measurement error has two aspects: precision and bias. Counting all salmon past a point, say at a fish ladder in a dam, for 24 hours a day, 7 days a week produces a number known without measurement error (barring the odd careless episode). Systematically counting all salmon past the same point every other hour throughout each day, 7 days a week produces an unbiased estimate of passage with an estimable variance that represents sampling error. Counting in an unsystematic fashion (i.e., only during daylight hours) will most likely produce a biased estimate of passage if reliable methods are unavailable to account for different rates of movement throughout the day.

1.5.1 Bias

Bias in estimates of spawning abundance or production will make the stock appear either more or less productive than it really is. If there is some independent information available (i.e., later studies showed that more salmon usually pass up the ladder at night than during the day), the direction of bias could be anticipated and adjustments made. Calibrating counts from aerial surveys with counts at a weir and against estimated abundance from mark-recapture experiments in the Stikine River is an example of such an adjustment. Without an adjustment, there is considerable danger of bias in estimates of S_{MSY} from statistical analysis; ending the analysis should be given serious consideration.

Bias is also involved with "outliers." Occasionally, one or more estimates of spawning abundance or production is atypical when compared to the others. Only if there is some functional reason to believe that this difference is due to failure in a stock assessment program, such as missing the peak of spawning in an aerial survey or a flash flood washing away many of the redds before they could be counted should these data be discounted. Barring any such independent information on a failure, the "outlier" should be included in the stock-recruit analysis.

1.5.2 Sampling Error

Whether expressed or not, measurement error follows from sampling. If estimated sampling variances are available, they should be combined to estimate variances for estimates of spawning abundance and production. When estimates of production and spawning abundance are independent (coming from separate sampling programs in different years), calculating statistics by year class is a matter of summing estimates by age and/or sex across years within a brood line. Below are a few helpful equations to estimate the statistical variance of alternative combinations of independent variables:

$$z = \sum_i c_i x_i \qquad v(z) = \sum_i c_i^2 v(x_i)$$

$$z = yx \qquad v(z) = y^2 v(x) + x^2 v(y) - v(x)v(y)$$

$$z = \frac{y}{x} \qquad v(z) \cong z^2 \left[\frac{v(y)}{y^2} + \frac{v(x)}{x^2}\right] = z^2\left[cv^2(y) + cv^2(x)\right],$$

where x and y are variables, c a constant, and $cv(\ldots)$ is the coefficient of variation. The first terms in a Taylor series expansion (the delta method described in Seber, 1982:7-9) can be used to approximate variance for any $z = g(x_1, x_2, \ldots x_n)$ such that:

$$v(z) \cong \sum_i^n v(x_i)\left[\frac{\partial g}{\partial x_i}\right]^2 + 2\sum_i^n \sum_{j>i}^n cov(x_i, x_j)\left[\frac{\partial g}{\partial x_i}\frac{\partial g}{\partial x_j}\right].$$

For instance, take the log-transformed variable $\ln(\hat{S}_y)$ where $\hat{S}_y$ is the estimated spawning abundance for year class y. Noting that the first partial derivative of $\ln(\hat{S}_y)$ is $\hat{S}_y^{-1}$:

$$v[\ln(\hat{S}_y)] \cong v(\hat{S}_y)\hat{S}_y^{-2} = cv^2(\hat{S}_y).$$

The same relationship holds for log-transformed estimates of production:

$$v[\ln(\hat{R}_y)] \cong v(\hat{R}_y)\hat{R}_y^{-2} = cv^2(\hat{R}_y).$$

When two estimates are based on independent sampling programs (estimated in different year and/or in different sampling programs), $cov(x_i, x_j) = 0$.

<u>Estimated Spawning Abundance:</u> Estimated variance for estimated spawning abundance has a two-stage structure with annual variation among the S plus sampling error for each estimate $\hat{S}_y$. If estimates of spawning abundance are to be log-transformed (as is usually the case):

$$V[\ln(\hat{S})] = V[\ln(S)] + \sigma_u^2,$$

where $V[\ln(S)]$ is the true variance for the <u>actual</u> spawning abundance over the years, $V[\ln(\hat{S})]$ the true variance of *estimated* spawning abundance over the years, and σ_u^2 represents the actual sampling error. Sampling error in estimated spawning abundance will bias parameter estimates, making the stock look more productive than it is ($\hat{S}_{MSY} < S_{MSY}$). Estimates of variance in <u>estimated</u> spawning abundance over the years and of sampling error are respectively:

$$v[\ln(\hat{S})] = \frac{\Sigma[\ln(\hat{S}_y) - \overline{\ln(\hat{S})}]^2}{n-1} \qquad \hat{\sigma}_u^2 = \frac{\sum cv^2(\hat{S}_y)}{n}. \qquad 1.1a, b$$

The estimate $\hat{\sigma}_u^2$ is based on a log-normal model of measurement error $\hat{S}_y = S_y \exp(u_y)$ where $u_y \sim N(0, \sigma_u^2)$. An estimate of the variance in the <u>actual</u> spawning abundance across years is therefore $v[\ln(S)] = v[\ln(\hat{S})] - \hat{\sigma}_u^2$. If the estimated sampling error $\hat{\sigma}_u^2$ represents:

- 10% or less of $v[\ln(\hat{S})]$, sampling error in spawning abundance may be ignored in further analysis;
- between 10% and 25%, the statistical stock-recruit analysis should be corrected for sampling error in spawning abundance (see Quinn and Deriso 1999:section 3.2.3); or
- greater that 25% of $v[\ln(\hat{S})]$, a statistical stock-recruit is of doubtful value.

Not knowing the sampling error does not make it disappear. If possible, sampling variances should be estimated. However, if sampling variances can not be calculated, there may be some reason to believe sampling error is <10% or >25% of $v[\ln(\hat{S})]$. In the former situation, statistical stock-recruit analysis should proceed; in the latter, it should end.

Estimated Production: Estimates of production also have measurement error. If sampling variances are known, an estimate of measurement error is:

$$\hat{\sigma}_v^2 = \frac{\sum cv^2(\hat{R}_y)}{n}. \qquad 1.2$$

The estimate $\hat{\sigma}_v^2$ is based a log-normal model of measurement error $\hat{R}_y = R_y \exp(v_y)$ where $v_y \sim N(0,\sigma_v^2)$.

Measurement error from estimating production affects estimates of S_{MSY} by potentially producing heteroscedastic (unequal) conditional variances and by confounding estimates of process error. Regression is based on homogenous conditional variances for the dependent variable. Dissimilarity in measurement error across estimates of production arising from annual vagaries in sampling programs would violate this assumption. The remedy would be a weighted regression (see Section 1.6). Fortunately, sampling effort is often similar from year to year, so the relative sampling error in estimates of production (the *cvs*) are often similar across years as well. For this reason, weighted regression is usually not needed.

Measurement error in the dependent variable is also confounded with process error. Estimates of S_{MSY} should be corrected for the presence of process error, but not for measurement error (see Section 1.7). An appropriate correction requires an estimate of measurement error or its ratio to process error. Not having this information does not make the biasing effect of measurement error disappear. If estimates of measurement error are not available, there should be some rationale other than ignorance to show why estimates of S_{MSY} were corrected for process error in the analysis.

1.6 Regression

As mentioned earlier, statistically regressing production against spawning abundance through a mathematical model is the traditional approach to estimating S_{MSY}. To do so successfully requires that circumstances meet the criteria on contrast specified in Section 1.4 and measurement error in Section 1.5. Statistical conditions for linear regression must also be met along with having enough data (>15 year classes represented) for parameter estimates from the analysis to be unbiased and reasonably precise. If regression is inappropriate, some inference concerning S_{MSY}

may be obtained via ad hoc methods based on the data at hand or from ancillary data. Maximum production of adults or smolt abundance given a spawning abundance, respectively, are examples of information that might provide some insight on the value of S_{MSY}. Because ad hoc methods are tailored to circumstances, they are not readily described in general. Since conditions for statistical regression of production against spawning abundance are usually met for chinook salmon along the Pacific coast, descriptions in the remainder of this section and the next two sections are restricted to topics concerning model-based statistical regression.

1.6.1 The Model

The stochastic two-parameter exponential model (Ricker's model) with multiplicative, log-normal error is the model of first choice:

$$R_y = \alpha S_y e^{-\beta S_y} \exp(\varepsilon_y),$$

where R_y is the production by year class y, S_y is the number of spawners that produced them, α is the density-independent parameter, β the density-dependent parameter, and ε_y represents log-normal process error with mean 0 and variance σ^2_ε. This model is most plastic of the common two-parameter models and more parsimonious than its three-parameter competitors. The linear form of Ricker's model has the difference of two logs as the dependent variable, $\ln(R_y) - \ln(S_y) = \ln(\alpha) - \beta S_y + \varepsilon_y$. When actual values of R_y and/or S_y are unavailable, as is almost always so, their estimates are used to produce the resulting adaptation:

$$\ln(\hat{R}_y) - \ln(\hat{S}_y) = \ln(\alpha) - \beta\hat{S}_y + r_y, \qquad 1.3$$

where $r_y = \varepsilon_y + u_y + v_y$ with u_y and v_y expressions of measurement error. The residuals $r_y \sim N(0, \sigma^2_r)$ where $\sigma^2_r = \sigma^2_\varepsilon + \sigma^2_{uv}$. The estimate $\hat{\sigma}^2_r$ is the mean square error in the regression [or can be calculated as the variance of the residuals times the quantity $(n-1)/(n-k)$ where k is the number of estimated parameters in the regression]. Measurement error σ^2_{uv} can be approximated through the delta method:

$$\hat{\sigma}^2_{uv,y} = v[\ln(\hat{R}_y) - \ln(\hat{S}_y)] = cv^2(\hat{R}_y) + cv^2(\hat{S}_y), \qquad 1.4a$$

$$\hat{\sigma}^2_{uv} = \frac{\sum \hat{\sigma}^2_{uv,y}}{n}. \qquad 1.4b$$

Sometimes the addition of covariates to the model will strengthen the estimated relationship between production and spawning abundance. For instance, production can be simultaneously regressed against both spawning abundance and survival rates (or their indicies). Any number of covariates can be added in a multiplicative fashion to Ricker's model as a competitive factor in establishing production:

$$\ln(\hat{R}_y) - \ln(\hat{S}_y) = \ln(\alpha) - \beta\hat{S}_y + \gamma_i X_{i(t)} + r_y \qquad 1.5$$

where $X_{i(t)}$ is the value of the ith covariate in year t and γ_i is the corresponding parameter. In such a formulation, the X_i represent either instantaneous rates or indices. For example, when $X_{(t)} \equiv M_{y(t)}$, an index of marine survival rates, the covariate is the log transform $\ln[M_{y(t)}]$. If there is a strong correlation between covariates and production, and a weak correlation between spawning abundance and these covariates, the estimated relationship between production and spawning abundance will be strengthened. If not, adding the covariates to the model will actually weaken the estimated relationship.

The formulation above is relevant when covariates play no density-dependent role in the production relationship. If there is an interaction between a covariate and spawning abundance in the regression, the covariate is part of the density-dependence. For instance:

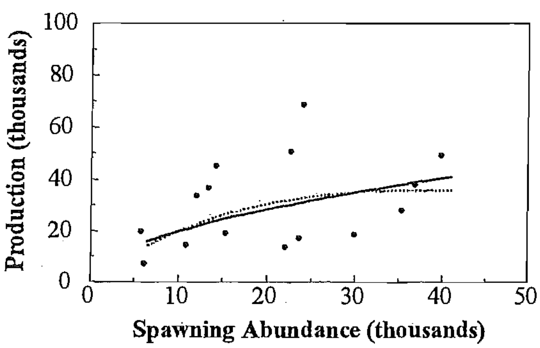
$$\ln(R_y) - \ln(S_y) = \ln(\alpha) - \beta S_y + \gamma \ln[M_{y(t)}] + \lambda S_y \ln[M_{y(t)}] + \varepsilon_y,$$

when the covariate represents an index of marine survival rates. The parameter λ here represents the "density-dependent" relationship between spawning abundance and marine survival rates arising from competition or predation. Because detecting such "density-dependent" covariates in regression analysis is unlikely given the usual precision and contrast in data, this model was not developed further.

If there is significant sampling error in estimates of spawning abundance, but not so much as to render the stock-recruit analysis too risky to complete, a regression on a simpler approximate model can be corrected to produce unbiased estimates of S_{MSY}. If production has a general positive trend with spawning abundance, as in the nearby figure, a power function:

$$R_y = \alpha' S_y^{\beta'} \exp(\varepsilon_y), \qquad 1.6$$

can be a good approximation to Ricker's model. The data displayed in the figure are for the population in the Stikine River with the dashed line representing Ricker's model and the solid line a power function. The linearized form of this power function is:



$$\ln(\hat{R}_y) = \ln(\alpha') + \beta' \ln(\hat{S}_y) + r_y, \qquad 1.7$$

where $r_y = \varepsilon_y + \nu_y$ and $\sigma_r^2 = \sigma_\varepsilon^2 + \sigma_\nu^2$. Unbiased estimates of parameters can be obtained with the following equations:

$$\hat{\beta}' = \frac{m_{XY}}{m_{XX} - \hat{\sigma}_u^2} = \frac{(n-1)^{-1}\sum_y^n (\ln \hat{S}_y - \overline{\ln \hat{S}})(\ln \hat{R}_y - \overline{\ln \hat{R}})}{(n-1)^{-1}\sum_y^n (\ln \hat{S}_y - \overline{\ln \hat{S}})^2 - \hat{\sigma}_u^2} \quad 1.8$$

$$\ln(\hat{\alpha}') = \overline{\ln \hat{R}} - \hat{\beta}' \overline{\ln \hat{S}} \quad 1.9$$

Procedures to estimate the variance-covariance matrix for $\ln(\hat{\alpha}')$ and $\hat{\beta}'$ and to calculate values for $\ln(R_y)$ and $\ln(S_y)$ corrected for sampling error can be found in Fuller (1987:13-26).

The model should also be adapted if data are demonstrably auto-correlated. Unbiased estimates from statistical regression are based on the independent variable truly being independent. This condition is not met for stock-production data because these data are comprised of two related parallel time series. If autocorrelation within these two series is weak (undetectable), fitting the models above can produce reasonably unbiased parameter estimates. The approach is to fit the appropriate model above to the data, then to subject results to tests to detect autocorrelation (these tests are described in Section 1.6.2). If autocorrelation is significant, the appropriate model is adapted to incorporate detected autocorrelation.

These adaptations and their interpretations are not trivial and are described in their own section (2.7).

1.6.2 Fitting the Model to Data

Parameters α and β and parameters for any covariates should be estimated with statistical software, such as SAS™, SYSTAT™, MINITAB™, S-Plus™, etc. which (should) have the following options and diagnostics:

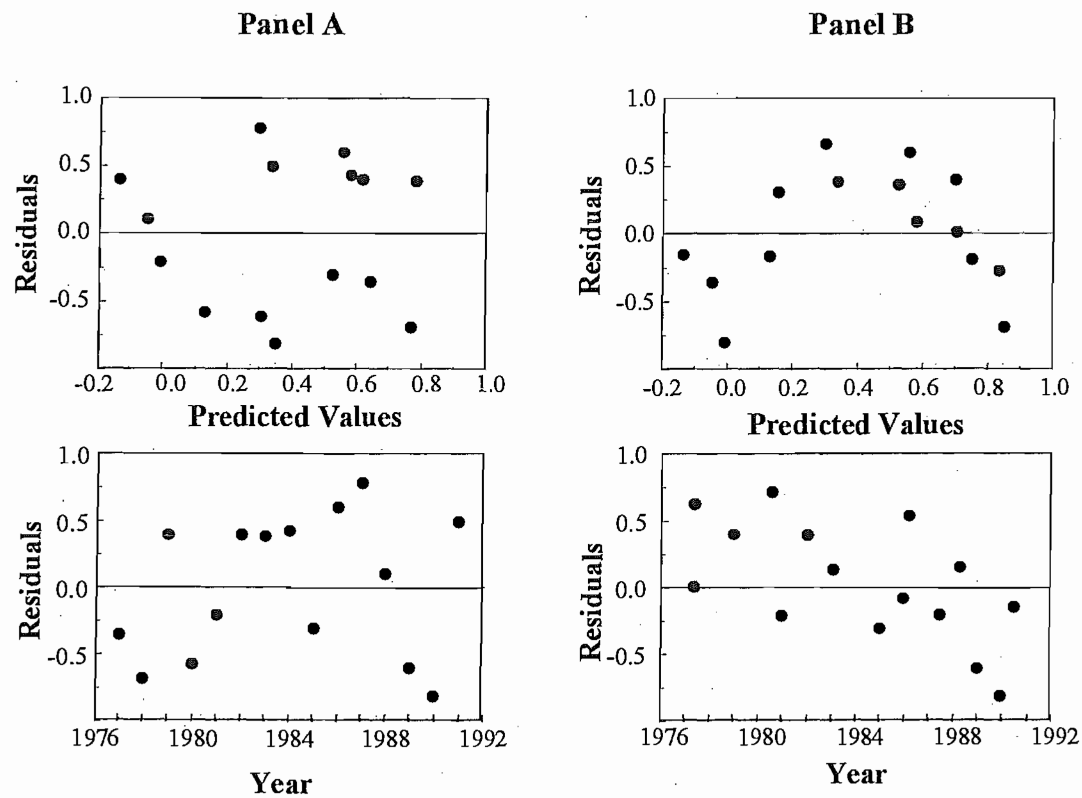
- possibilities for weighted regression (option);
- output residuals (diagnostic); and
- options to calculate autocorrelation and partial autocorrelation functions (diagnostic);

Weighted regression: When conditional (sampling) variances for the dependent variable are heterogeneous across the range of the independent variable, dependent variables should be weighted in the regression. Such an instance would occur if sampling effort changed dramatically over the years, such as beefing up catch sampling as time progressed or inclement weather one year dramatically reduced sampling effort. The weights w_y are the reciprocals of the estimated sampling variances for the dependent variable, here $v[\ln(\hat{R}_y) - \ln(\hat{S}_y)] = \hat{\sigma}_{uv}^2$.

Be aware that some software assumes that you have flipped the variances to produce the weights while other programs will flip the variances for you. Know the difference, and act accordingly. Weighted regression will often improve the fit of the model (R^2, the coefficient of determination) and can dramatically change parameter estimates, however, if log-transformed variances are similar across all years classes, these improvements and changes will be negligible.

Residuals. Differences between observed values for the dependent variable and their predictions from the regression can be used in diagnostic tests, usually in the form of plots. If these residuals are randomly spread regardless of predicted values (which is usually the case), there is no evidence for influence of spawning abundance beyond that expressed in Ricker's two-parameter model. Panel A, top plot in the figure below demonstrates a pattern of residuals plotted against predicted production where no higher order influence of spawning abundance is indicated. Panel B, top plot demonstrates an atypical situation where there should be an additional quadratic expression for spawning abundance in the stock-recruit relationship. A stock-recruit relationship with a disequilibrium point would have a residual plot similar to that demonstrated in Panel B, top plot. This pattern is atypical because density-independent processes usually hide such subtleties in determining year-class strength of salmon.

A plot of residuals against year can reveal serial correlation in the original time series of production. If no serial correlation is indicated, as in Panel A, bottom plot in the figure below, the original time series is said to be stationary, which is a good thing. If there is a trend in the residuals, as in Panel B, bottom plot, the original time series of production is non-stationary, and we are left with a philosophical dilemma (see discussion in Section 2.7 on non-stationary data).



Autocorrelation. Autocorrelation functions (ACFs) and partial autocorrelation functions (PACFs) of residuals are tools used to detect the potential for time-series bias in estimates (see the next section). Both functions are typically plotted as bar graphs with each bar corresponding to ascending number of lags in the data. Each bar in an ACF plot represents the estimated correlation between individual residuals lagged k years among all possible pairs z_y, z_{y-k} where z_y is the residual for year class y. For the left-most bar, k=1; for the next bar to the right, k=2; etc. Often there are monotonically increasing lines on the top and bottom of each ACF plot corresponding to 95% confidence limits. If the bars cross these lines, a statistically significant autocorrelation is indicated. Partial autocorrelation functions are displayed in the same fashion, only the PACFs are partial regression coefficients δ_{kk} in the representation:

$$z_y = \delta_{k1} z_{y-1} + \delta_{k2} z_{y-2} \ldots + \delta_{kk} z_{y-k} + a_y,$$

where a_y is the "white-noise" error.

1.7 Correcting for Autocorrelation

When residuals from a stock-production regression are demonstrably dependent through time (auto-correlated) as detected with hypothesis tests, the traditional approach has the potential for time-series bias in estimates of S_{MSY}. Auto-correlated residuals can arise from three processes: non-stationary, auto-regressive, and moving-average. An auto-regressive process reflects cycles in the environment that influence survival rates; a moving-average process reflects that future production and spawning abundance as a function of past production and spawning abundance; and a nonstationary process indicates that production is changing according to a yet unfinished trend.

1.7.1 Types of Autocorrelation

Time-series bias from a nonstationary process is the most pernicious in that it is often uncorrectable. There are methods, such as differencing, that can be used to "stabilize" the data for statistical analysis, however, this correction is usually not meaningful. Stock-recruit analysis is predicated on the data collected in the past being representative of the future. A nonstationary time series of production indicates a trend that has not yet run its course. In short, past data do not reflect future conditions. No statistical analysis of nonstationary data on spawning abundance and production can result in an accurate estimate of optimal production for the future. Nonstationary time-series result when the series is too short relative to important natural variation in the environment, or to longer-term (agricultural development) or to immediate (dam construction) degrading of habitat. A nonstationary process is indicated when production follows an obvious monotonic trend across the years.

One possible exception arises when there is a trend in exploitation rates for stocks that are subject to significant harvest as immature fish. For instance, production will appear to rise with rising exploitation rates because immature fish will be caught when they are younger and more numerous. If exploitation rates slacken, the converse is true. In this instance, there are two methods for transforming the data: calculation of adult equivalent mortalities and differencing. The first method has been described above (Section 1.3 and Appendix A2.). In the second, two new time series, $\{\nabla\mathbf{R}\}$ and $\{\nabla\mathbf{S}\}$, can be created from the original data through first differencing, that is by subtracting consecutive data:

$$\nabla R_y = R_y - R_{y-1} \quad \nabla S_y = S_y - S_{y-1}$$

There should be some independent evidence of increasing (or decreasing) exploitation rates from direct estimates of rates or trends in fishing effort mimicking the trend in production. If more than first-order differencing is needed (the equations immediately above), there is probably more than just the trend in exploitation rates at work, and the usefulness of the data would again be in doubt. For stocks only exploited as adults, trends in exploitation rates are not the cause of nonstationary data.

An auto-regressive process arises when process error is correlated across years. If an auto-regressive process is ignored, estimates of parameters in the statistical stock-recruit analysis will be little affected, however, the fit of the stock-recruit model to the data will often appear better than it really is. This phenomenon has been called "spurious regression." Ironically, correcting for this type of time-series bias will often degrade precision in estimates (and rightly so).

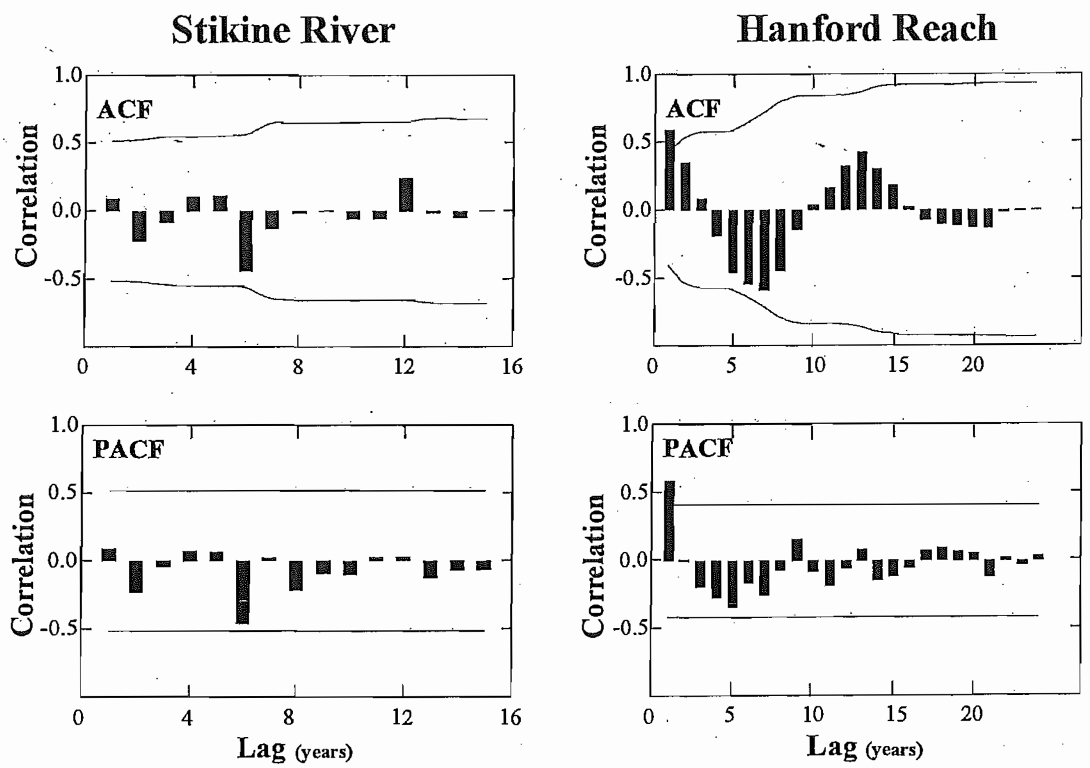
A moving-average process often results when this year's production largely becomes the spawning abundance for the next generation. If ignored, time-series bias from moving-average processes can be severe, usually making the population look more productive than it is ($\hat{S}_{MSY} < S_{MSY}$). Fortunately, this type of time-series bias should be uncommon for chinook salmon because production by a year class is spread out over several years and is heavily influenced by density-independent factors in the environment.

1.7.2 Model Adaptations

Auto-regressive and moving-average processes can be modeled separately or together, then fit to data to remove the potential for time-series bias. The nature of the appropriate model can be determined through inspection of autocorrelation functions (ACFs) and partial autocorrelation functions (PACFs) of the residuals from standard fits of Ricker's linearized model or a linearized power function. The table of diagnostics below was derived from Abraham and Ledolter (1983:250):

MODEL:	**ACF:**	**PACF:**
AR(p)	ACFs trail off exponentially or cyclically after p significant lags	PACFs cut off after p significant lags
MA(q)	PACFs cut off after q significant lags	PACFs trail off exponentially or cyclically after q significant lags
ARMA(p,q)	ACFs trail off exponentially or cyclically after p significant lags	PACFs trail off exponentially or cyclically after q significant lags

The following figure demonstrates how ACFs and PACFs can be used to determine the appropriate model. Note for the Hanford-Reach residuals only the ACF with lag one generation is significant with the other ACFs cycling while exponentially declining at higher lags. The PACF of lag one generation is significant for this population as well, with the remaining PACFs non-significant and bouncing around 0 with little pattern. This situation denotes an ARMA(1,0) model to account for autocorrelation. In contrast, Stikine River residuals have no significant ACFs or PACFs; no autocorrelation is indicated.



Using the methods described in (Noakes et al. 1987) and Pankratz (1991), Ricker's linearized production model can be generalized as an ARMA(p,q) model:

$$\ln(R_y) - \ln(S_y) = \alpha - \beta S_y + a_y(1-\theta(B))^1(1-\phi(B))^{-1}, \qquad 1.10$$

where $\phi(B)$ and $\theta(B)$ are polynomial functions of B, the backshift operator (returns the statistic from the previous year):

$$\phi(B) = 1 - \phi_1 B - \phi_2 B^2 - \cdots - \phi_q B^p$$
$$\theta(B) = 1 - \theta_1 B - \theta_2 B^2 - \cdots - \theta_q B^q,$$

and where $\{\phi\}$ are auto-regressive parameters, $\{\theta\}$ are moving-average parameters, and a_y is an independent error distributed with mean 0 and variance σ_a^2. Generally a lag of one year ($p = 1$ and/or $q = 1$) is sufficient to describe autocorrelation in residuals for salmon. The AR(1) model is:

$$\ln(R_y) - \ln(S_y) = \ln(\alpha) - \beta S_y + a_y(1-\phi_1 B)^{-1} \qquad 1.11$$

or equivalently, after multiplying both sides of the equation by 1 - ϕ_1B:

$$\ln(R_y) = (1-\phi_1)\ln(\alpha) + \phi_1 \ln(R_{y-1}) + \ln(S_y) - \beta S_y - \phi_1 \ln(S_{y-1}) + \phi_1\beta S_{y-1} + a_y \qquad 1.12a$$

Time series software, such as SAS ™ ETS, Autobox™, etc. can be used to provide the fits. When variables are not known, but are estimated, substitute $\hat{R}_y \to R_y$ and $\hat{S}_y \to S_y$ in the formulations above:

$$\ln(\hat{R}_y) = (1-\phi_1)\ln(\alpha) + \phi_1 \ln(\hat{R}_{y-1}) + \ln(\hat{S}_y) - \beta\hat{S}_y - \phi_1 \ln(\hat{S}_{y-1}) + \phi_1\beta\hat{S}_{y-1} + r_y \qquad 1.12b$$

Sampling error in the dependent variable can still be handled with weighted regression if needed. Sampling error in the independent variable could be more of a problem, however, observing strong autocorrelation and large measurement error together is not likely. Measurement error will tend to obscure autocorrelation within a time series as well as cross-correlation between them.

A fit of the model above to data from the Hanford Reach demonstrates well the kind of time-series bias to be expected from an auto-regressive process. The estimate of β in the traditional fit of Ricker's linearized model is 0.0000284; the estimate with the ARMA(1,0) model is 0.0000252. *Student's t* for the traditional fit was 4.65 while that for the ARMA(1,0) was 2.49; still significant, but less precise. From the traditional fit, $\hat{S}_{MSY} = 31{,}103$; from the ARMA(1,0) model 33,484.

1.8 Estimating S_{MSY} and U_{MSY}

1.8.1 Optimal Spawning Abundance

For the stochastic form of Ricker's model, an estimate $\hat{S}_{MSY}$ can be obtained by iteratively solving the transcendental function:

$$1 = (1 - \hat{\beta}\hat{S}_{MSY})\exp(\ln\hat{\alpha})\exp(-\hat{\beta}\hat{S}_{MSY})\exp(\hat{\sigma}_\varepsilon^2/2)\,. \qquad 1.13$$

If spawning abundance is expressed only in females while production represents both sexes, the equation to estimate S_{MSY} becomes:

$$1 = (1 - \hat{\beta}\hat{S}_{MSY})\exp(\ln\hat{\alpha})(1+\tau)^{-1}\exp(-\hat{\beta}\hat{S}_{MSY})\exp(\hat{\sigma}_{\varepsilon}^{2}/2) \qquad 1.14$$

where τ is the expected ratio of males to females in the population. The same equations are used when Ricker's model has been transformed into an ARMA model. Because $\{\phi\}$ and $\{\theta\}$ are involved solely in the error term in the ARMA model, these parameters are nuisance parameters, and their estimates are unused in a solution to estimate S_{MSY}.

The equations above to estimate S_{MSY} and those below contain a correction for process error in the stock-production relationship [$\exp(\hat{\sigma}_{\varepsilon}^{2}/2)$]. Without it, the estimate of S_{MSY} will represent the maximum median value of production, not the maximum mean (Hilborn 1985). Because mean production is greater than median production log-normal process error, the former represents the true expectation in yield from a given spawning abundance. Application of this correction will increase the estimate of S_{MSY}.

Methods to estimate σ_{ε}^{2} depend on the presence and knowledge of measurement error in the data. In the rare case where R_y and S_y are known, $\hat{\sigma}_{\varepsilon}^{2} = \hat{\sigma}_{r}^{2}$ where $\hat{\sigma}_{r}^{2}$ is the mean square error in the fitted model. However, when $\hat{R}_y \to R_y$ and/or $\hat{S}_y \to S_y$, $\hat{\sigma}_{\varepsilon}^{2} < \hat{\sigma}_{r}^{2}$, and the correction $\exp(\hat{\sigma}_{r}^{2}/2)$ will "*over correct*" such that $S_{MSY} < \hat{S}_{MSY}$. If sampling variances are available, the appropriate correction factor can be found through subtraction. For Ricker's model, $\hat{\sigma}_{\varepsilon}^{2} = \hat{\sigma}_{r}^{2} - \hat{\sigma}_{uv}^{2}$. If the ratio of measurement error to process error λ is known or estimated, $\hat{\sigma}_{\varepsilon}^{2} = \hat{\sigma}_{r}^{2}/(1+\hat{\lambda})$. For Ricker's model, $\lambda = \sigma_{uv}^{2}/\sigma_{\varepsilon}^{2}$.

If the traditional production model contains covariates, such as an index of marine survival (production), estimates of S_{MSY} can be found by iteratively solving:

$$1 = (1 - \hat{\beta}\hat{S}_{MSY})\exp(\ln\hat{\alpha})\exp(\textstyle\sum_i \hat{\gamma}_i \bar{X}_i)\exp(-\hat{\beta}\hat{S}_{MSY})\exp(\hat{\sigma}_{\varepsilon}^{2}/2), \qquad 1.15$$

where $\bar{X}_i$ is the arithmetic or geometric means (medians, n-tiles) of the ith covariates over the years in the data series. This equation is modified by multiplying the right-hand sides by $(1+\tau)^{-1}$ if females represent spawning abundance. When $0 < \ln(\alpha) < 3$, $\hat{S}_{MSY}$ can be approximated without involving iteration. Hilborn and Walters (1992:271-2) published the following empirical approximation:

$$\hat{S}_{MSY} \cong \frac{\ln\hat{\alpha} + \hat{\sigma}_{\varepsilon}^{2}/2}{\hat{\beta}}[0.5 - 0.07(\ln\hat{\alpha} + \hat{\sigma}_{\varepsilon}^{2}/2)]. \qquad 1.16$$

This approximation holds only when spawning abundance and production are measured in the same units and no covariates are included in the analysis.

If a log-transformed power function has been used to approximate the linearized Ricker model:

$$\hat{S}_{MSY} = \sqrt[\hat{\beta}'-1]{(\hat{\alpha}'\hat{\beta}')^{-1}}\,, \tag{1.17}$$

where $\hat{\alpha}' \cong \exp[\ln(\widehat{\alpha'}) + \hat{\sigma}_\varepsilon^2/2]$. The correction for process error is included in this and the next model as well, only measurement error in the correction is represented only by $\hat{\sigma}_v^2$ with $\hat{\sigma}_\varepsilon^2 = \hat{\sigma}_r^2 - \hat{\sigma}_v^2$ or $\hat{\sigma}_\varepsilon^2 = \hat{\sigma}_r^2/(1+\hat{\lambda})$ with $\lambda = \sigma_v^2/\sigma_\varepsilon^2$. If only females represent spawning abundance:

$$\hat{S}_{MSY} = \sqrt[\hat{\beta}'-1]{\frac{1+\tau}{\hat{\alpha}'\hat{\beta}'}}\,. \tag{1.18}$$

1.8.2 Estimated Variance

The estimated variance $v(\hat{S}_{MSY})$ and 90% confidence intervals for $\hat{S}_{MSY}$ can be calculated through non-parametric bootstrapping of residuals from the regression (see Efron and Tibshirani 1993:111-5). Residuals are calculated as differences between observed and predicted values for the R_y:

$$\zeta_y = R_y - \hat{\mathrm{E}}[R_y]\,, \tag{1.19}$$

where $\hat{\mathrm{E}}[R_y]$ is the predicted value of production for year class y. Independent variables here would be spawning abundance and any covariates. Note that if production is not known, but is estimated, $\hat{R}_y \to R_y$ in the equation above. A new set of dependent variables (production) are then generated with the residuals from the original regression in general and specific predictions of production:

$$\tilde{R}_y = \zeta_y^* + \hat{E}[R_y]\,, \tag{1.20}$$

where the ζ_y^* are drawn randomly with replacement from the original vector ζ of the original residuals. In this manner, a new value for production $\tilde{R}_y$ is simulated for each year class in the data series creating a new data set comprised of the original values for the independent variables (spawning abundance and covariates) and simulated values for production. The $\tilde{R}_y$ are then regressed against the original values of the independent variables to produce a new, simulated vector of parameter estimates $\tilde{\mathbf{p}}$ (including $\tilde{\sigma}_\varepsilon^2$). Say that K such vectors are drawn, and $b = 1 \to K$. For each new vector $\tilde{\mathbf{p}}_b$, the simulated parameter estimates are plugged into the appropriate relationship above (Equation 1.13-18), then the relationship solved for $\tilde{S}_{MSY(b)}$. Over K simulations, the estimated variance for $\hat{S}_{MSY}$ is (from Efron and Tibshirani 1993:47):

$$v(\hat{S}_{MSY}) = \frac{\sum_{b=1}^{K} (\tilde{S}_{MSY(b)} - \bar{S}_{MSY})^2}{K-1} \qquad 1.21$$

where $\bar{S}_{MSY} = K^{-1}\sum_{b=1}^{K} \tilde{S}_{MSY(b)}$. The difference between $\hat{S}_{MSY}$ and $\bar{S}_{MSY}$ is an indication of statistical bias in the former statistic (note this statistical bias is assumed to arise only from process error in the regressions of production against spawners). When estimated through simulation, $v(\hat{S}_{MSY})$ represents a mean square error, that is, it represents both the actual variance of $\hat{S}_{MSY}$ plus statistical bias in the estimate. The original estimate $\hat{S}_{MSY}$, not $\bar{S}_{MSY}$, is the preferred estimate of spawning abundance that produces *MSY*, even though the former statistic contains some uncorrected bias (see Efron and Tibshirani 1993:138).

Confidence intervals about $\hat{S}_{MSY}$ can be estimated from the *K* simulations with the percentile method (Efron and Tibshirani 1993:124-126). The *K* values of $\tilde{S}_{MSY}$ are sorted in ascending order. For a 90% interval, the lower bound is the value of $\tilde{S}_{MSY}$ at the [(0.05)*K*+1]th place down the list; the upper bound is the [(0.95)*K*-1]th value. Confidence intervals of different sizes can be determined from the same list using the same procedure.

Another type of non-parametric bootstrap is to resample the original data with replacement to produce new sets of parameter estimates, $\tilde{\mathbf{p}}$ (see Efron and Tibshirani 1993:113). This approach is not recommended. Original stock-production data and covariates are not randomly selected as implied in this resampling scheme, but are time series data with chronological structure. Resampling the original data would destroy that structure.

When S_{MSY} can be estimated directly with the Hilborn's approximation (Equation 1.18), there is the option of estimating $V(\hat{S}_{MSY})$ through maximum-likelihood calculations and confidence intervals with profile likelihoods. Bootstrap simulation can still be used on this approximation, however, the simulations require considerably more calculation.

A statistical confidence interval about $\hat{S}_{MSY}$ is an expression of the uncertainty in the estimate and only by chance would be an expression of the productivity in the stock. Eggers (1993) showed through simulation based on management of sockeye salmon (*O. nerka*) fisheries that 90% of MSY could be obtained so long as spawning abundance fluctuated within the range of 80 to 160% of S_{MSY}. Such a range probably exists for all salmon stocks, including stocks of chinook salmon. This "productivity range" would remain even if we attained perfect knowledge of S_{MSY} and the confidence interval about $\hat{S}_{MSY}$ shrunk to zero.

1.8.3 Optimal Exploitation Rates

Calculation of the exploitation rate U_{MSY} that is associated with S_{MSY} is less model-dependent once S_{MSY} has been estimated. For Ricker's model:

$$\hat{U}_{MSY} = \hat{\beta}\,\hat{S}_{MSY} : \text{Ricker's Model.} \qquad 1.22$$

This relationship holds regardless of covariates, adjustments for different units, or for use of Hilborn's approximation. The equation to estimate optimal exploitation rates with the power function requires a "de-transformation" of one of the estimated parameters:

$$\hat{U}_{MSY} = 1 - \hat{\beta}' : \text{Power Function,} \qquad 1.23$$

where $\hat{\alpha}' \cong \exp[\ln(\hat{\alpha}') + \hat{\sigma}_{\varepsilon}^{2}/2]$. Variances $v(\hat{U}_{MSY})$ and confidence intervals for $\hat{U}_{MSY}$ can be estimated during the same simulations used to estimate $V(\hat{S}_{MSY})$. When U_{MSY} is estimated with Hilborn's approximation (Equation 1.18), there is the option of estimating $V(\hat{U}_{MSY})$ through maximum-likelihood calculations and confidence intervals with profile likelihoods.

2. SOUTHEAST ALASKA

2.1. Stock Specific Descriptions and Analysis

Annual CTC escapement assessments include 11 indicator stocks of chinook salmon that originate in streams of Southeast Alaska (SEAK) or in transboundary waters. The CTC concluded that biologically based escapement goals and associated ranges were currently established for eight of these stocks after review of agency reports summarizing analysis of stock-recruit relationships (CTC 1998). Agency reports associated with derivation of escapement goals for these eight stocks are summarized below. Rivers which produce chinook salmon in this region are identified in Figure 2.1.

Stock	Escapement Goal Range	Agency Report
Situk River	500 to 1,000 total large spawners	McPherson 1991
Unuk River	Index count of 650 to 1,400 large spawners[a]	McPherson and Carlile 1997
Chickamin River	Index count of 450 to 900 large spawners[a]	McPherson and Carlile 1997
Keta River	Index count of 250 to 500 large spawners[a]	McPherson and Carlile 1997
Blossom River	Index count of 250 to 500 large spawners[a]	McPherson and Carlile 1997
King Salmon River	120 to 240 total large spawners	McPherson and Clark *In Press*
Andrew Creek	650 to 1,500 total large spawners	Clark, McPherson, and Gaudet 1998
Alsek River	1,100 to 2,300 total spawners in the Klukshu River	McPherson, Etherton, and Clark 1998

[a] Mark-recapture studies from 1994 to 1998 on Behm Canal Rivers indicate that expansion factors are approximately 5 for the Unuk and Chickamin Rivers and about 4 for the Keta and Blossom Rivers; e.g., the escapement range for the Unuk River would approximately 3,250 to 7,000 total large spawners if expanded by this factor.

Two new agency reports for the Stikine and Taku stocks of chinook salmon are currently being developed. Information from these agency reports are summarized herein and provide the basis for biological escapement goals for chinook salmon spawning in the Stikine and Taku Rivers.

2.2. Stikine River

2.2.1 Summary

Year Classes	S_{MSY} Range	$\hat{S}_{MSY}$ (90% C.I.)	$\ln\hat{\alpha}$	$\hat{\beta}$	$\hat{U}_{MSY}$ (90% C.I.)
1977 - 1991	14,000 – 28,000	17,368 (11,838 – 39,907)	0.9591	0.00002676	0.46 (0.26 – 0.61)

An escapement goal range of 14,000 to 28,000 adult spawners (3-5 ocean-age) is recommended for this stock. This recommendation was accepted in the spring of 1999 by the Chinook Technical Committee, an internal review committee of ADF&G, and by the PSC's Transboundary Technical Committee. The Pacific Stock Assessment and Research Committee of CDFO declined to pass judgement on this range in deference to the decision by the Transboundary Technical Committee, but judged the estimated exploitation rate of 46% to be too high with 32% being a better rate. Statistics on harvest and spawning abundance by age group were used to estimate production which was regressed against estimated spawning abundance in accordance with Ricker's model. Measurement error in estimates of production and spawning abundance was incorporated into the analysis and was shown to be negligible relative to overall variation. Confidence intervals were estimated with the percentile method applied to the results of bootstrap simulations.

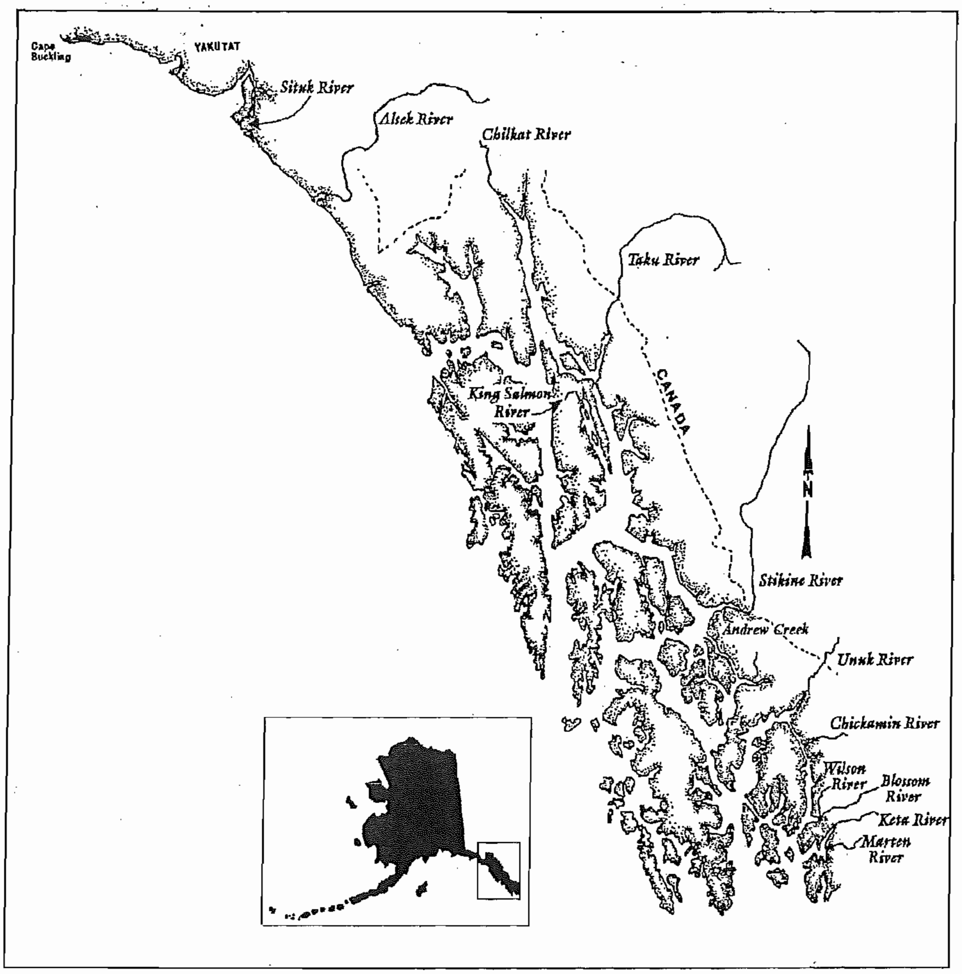


Figure 2.1. Location of selected chinook salmon systems in Southeast Alaska, Yakutat, and transboundary rivers.

Primary Reference: Bernard, D. R., S. A. McPherson, S. A., K. Pahlke, and P. Etherton. *In Prep.* Optimal Production of chinook salmon from the Stikine River. Alaska Department of Fish and Game, Sport Fish Division, Fishery Manuscript No. 99__, Anchorage.

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2.2.2. Stock Description and Stock Assessment Data

Watershed Description: The Stikine River is a large glacially influenced watershed of approximately 52,000 km^2 with its origin in British Columbia and its terminus in SEAK. Because of natural barriers to migration, spawning of chinook salmon is limited mostly to downstream tributaries, such as the Iskut, Tahltan, and Little Tahltan Rivers and to Beatty, Christina, and Verrett Creeks. The main-stem Stikine River is turbid from late spring through early fall.

Stock Description: Chinook salmon from the Stikine River are a "spring" run of salmon with adults spawning in Canada from late July to mid-September. Almost all juveniles rear for one year in freshwater after emergence and are yearling smolt. Chinook salmon from the Stikine River rear offshore away from troll, sport, and net fisheries in SEAK, then as they mature, return to the river through SEAK from late April through early July (Kissner and Hubartt 1986). A few (~100 to ~1,000) are caught in early summer openings of the SEAK troll fishery. Presently, the annual migration is targeted only in a marine sport fishery beginning in early May. A terminally located marine gillnet fishery starts in late June after about three quarters of the run has migrated into freshwater. Only a few thousand chinook salmon are harvested annually in these marine fisheries. In the Stikine River itself, a few hundred to a few thousand migrating chinook salmon are incidentally captured in Canadian commercial and aboriginal fisheries for sockeye salmon each year.

Hatchery Influence: Hatchery chinook salmon have not been released in this transboundary river nor in nearby waters. Strays from hatcheries have not been found on the spawning grounds and hence hatchery influence to this stock is considered nil.

Escapement Estimation Method: Starting in 1975, ADF&G began counting chinook salmon spawning in stretches of the Little Tahltan and Tahltan rivers by helicopter, with fixed schedules and protocols. Surveys were expanded in 1980 to cover parts of Beatty Creek. Only large, typically 3-ocean age and older fish were counted. Smaller chinook salmon were not counted because they could not be distinguished from other species. Beginning in 1985, the CDFO counted chinook salmon through a weir on the Little Tahltan River, installed on the downstream margin of the area surveyed from helicopter by ADF&G. CDFO counts were segregated into small and large chinook salmon to correspond to ADF&G aerial surveys. In 1996 and 1997, abundance of large spawners in the Stikine River was estimated with two-event, closed-population mark-recapture experiments and with a radiotelemetry study. The Tahltan First Nation, ADF&G, and CDFO cooperated in these studies. These three escapement databases were used to develop total spawner abundance estimates in the Stikine River for the years 1975–1997 (full details of estimation methodology can be found in Bernard et al. *In Prep.*).

Age and Sex Composition of Escapement: Chinook salmon returning to the Stikine River mature after one to five years at sea. Fish returning at a younger age (age 1.1 and 1.2) are almost exclusively males, while older fish (age-1.3, 1.4, and 1.5) are consistently about 50% females. Ages 1.3 and 1.4 dominate the annual spawning population; age-1.2 and -1.5 fish are uncommon (less than 5%). Only a few return as age 1.1 salmon. Details concerning age composition and sample sizes associated with annual estimated age compositions of the Stikine River stock of chinook salmon can be found in Bernard et al. (*In Prep*.).

Production. Because only mature chinook salmon in this population are believed to be vulnerable to fisheries, production estimates are based on terminal returns; no adjustment for harvesting immature fish (calculating adult equivalents) was needed. Spawning abundance by age and estimated annual harvests in terminal fisheries by age were used to develop estimates of total production from spawning populations in the years 1977-1991. Table 2.1 contains annual estimates of spawning abundance and estimated production from these escapements along with standard errors of estimates (see Bernard et al. *In Prep* for details concerning methodology).

2.2.3. Stock-Production Analysis

Measurement Error: Values of R_y (production of adults by year class y) and S_y (spawning abundance that produced them) are unknown for the Stikine River population; only estimates are available such that:

$$\hat{R}_y = R_y \exp(v_y),$$
$$\hat{S}_y = S_y \exp(u_y),$$

where v_y and u_y represent log-normal measurement error with means 0 and variances σ_v^2 and σ_u^2. Transforming the above relationships produces:

$$\ln(\hat{R}_y) = \ln(R_y) + v_y,$$
$$\ln(\hat{S}_y) = \ln(S_y) + u_y,$$

Over the years, variance in $\ln(\hat{S})$ has a two-stage structure with annual variation among the S plus measurement error for each estimate $\hat{S}_y$ (see Section 1.5.2):

$$V[\ln(\hat{S})] = V[\ln(S)] + \sigma_u^2$$

These variances are also unknown, but can be estimated:

$$v[\ln(\hat{S})] = \frac{\sum[\ln(\hat{S}_y) - \overline{\ln(\hat{S})}]^2}{n-1}$$

$$\hat{\sigma}_u^2 = \frac{\sum \hat{\sigma}_{u,y}^2}{n},$$

$$v[\ln(S)] = v[\ln(\hat{S})] - \hat{\sigma}_u^2,$$

where n is the number of year classes in the data. The estimates $\hat{\sigma}_{u,y}^2$ are related to the sampling variances listed as standard errors in Table 2.1. However, those sampling variances must be log transformed as were estimates $\hat{S}_y$ and $\hat{R}_y$. From the delta method (see Section 1.5.2):

$$\hat{\sigma}_{u,y}^2 = v[\ln(\hat{S}_y)] \cong v(\hat{S}_y)\hat{S}_y^{-2} = cv^2(\hat{S}_y)$$

Table 2.1. Estimated spawning abundance $\hat{S}_y$ and estimated production $\hat{R}_y$ along with their standard errors (*se*) and coefficients of variation (*cv*) for the 1977–1991 year classes of chinook salmon in the Stikine River.

Year Class	$\hat{S}_y$	$se(\hat{S}_y)$	$cv(\hat{S}_y)$	$\hat{R}_y$	$se(\hat{R}_y)$	$cv(\hat{R}_y)$
1977	11,445	1,865	16.3%	15,223	1,704	11.2%
1978	6,835	1,465	21.4%	7,520	936	12.4%
1979	12,610	2,704	21.4%	35,107	3,423	9.8%
1980	30,573	4,982	16.3%	19,438	1,744	9.0%
1981	36,057	7,731	21.4%	29,245	2,974	10.2%
1982	40,488	6,598	16.3%	51,568	5,219	10.1%
1983	6,424	1,377	21.4%	20,575	1,980	9.6%
1984	13,995	3,000	21.4%	38,284	3,322	8.7%
1985	16,037	2,392	14.9%	20,000	2,132	10.7%
1986	14,889	2,221	14.9%	47,132	4,331	9.2%
1987	24,632	3,674	14.9%	71,951	7,903	11.0%
1988	37,554	5,601	14.9%	39,733	4,167	10.5%
1989	24,282	3,622	14.9%	17,947	1,798	10.0%
1990	22,619	3,374	14.9%	14,659	1,195	8.2%
1991	23,206	3,461	14.9%	54,824	3,221	5.9%

For the population in the Stikine River, $v[\ln(\hat{S})] = 0.3352$ and $\hat{\sigma}_u^2 = 0.0316$. Thus, measurement error represents about 9% of overall variation in estimated spawning abundance.

For production, only the second-stage variance need be estimated because first-stage variance is a function of variation in spawning abundance and process. If sampling variance $\hat{\sigma}_{v,y}^2$ represents measurement error in $\hat{R}_y$, the overall second-stage estimate of variance is (see Section 1.5.2):

$$\hat{\sigma}_v^2 = \frac{\sum \hat{\sigma}_{v,y}^2}{n},$$

$$\hat{\sigma}_{v,y}^2 = v[\ln(\hat{R}_y)] \cong v(\hat{R}_y)\hat{R}_y^{-2} = cv^2(\hat{R}_y),$$

For the population in the Stikine River, $\hat{\sigma}_v^2 = 0.0097$. Estimated measurement error for the estimated log of the production-to-spawner ratio $\hat{R}_y/\hat{S}_y$ is:

$$\hat{\sigma}_{uv,y}^2 = v[\ln(\hat{R}_y) - \ln(\hat{S}_y)] = cv^2(\hat{R}_y) + cv^2(\hat{S}_y).$$

The average over all year classes is $\bar{\hat{\sigma}}_{uv}^2 = 0.0407$ for data from the Stikine River.

<u>Parameter Estimates:</u> Two models were used in the analysis: Ricker's exponential function $R_y = \alpha S_y \exp(-\beta S_y)\exp(\varepsilon_y)$ and Cushing's power function $R_y = \alpha' S_y^{\beta'} \exp(\varepsilon_y)$. The latter is an approximation to the former that allows incorporation of measurement error in spawning abundance in the model. The term ε_y represents process error in both models where $\varepsilon_y \sim N(0, \sigma_\varepsilon^2)$. Parameters were estimated for the linear form of Ricker's model $\ln(R_y) - \ln(S_y) = \ln(\alpha) - \beta S_y + \varepsilon_y$ (Table 2.2) with the computer program PROC REG supported by SAS™. Because there was little variation in the range of estimated variances for production (Table 2.1), weighted regression was not needed to stabilize conditional variances (Section 1.6.2). Plots of residuals against predicted values of the dependent variable indicated spawning abundance has no remaining predictive power (Figure 2.2); there was no evidence of autocorrelation among residuals (Figure 2.3). Predictions by the fitted, untransformed model and the original data are plotted in Figure 2.4.

$$\tilde{R}_y = \exp(\widehat{\ln\alpha})\hat{S}_y \exp(-\hat{\beta}\hat{S}_y)\exp(\hat{\sigma}_\varepsilon^2/2).$$

Table 2.2. Estimated parameters for regression on the log-linear transform of Ricker's model on estimates of production and spawning abundance of chinook salmon in the Stikine River.

$\widehat{\ln(\alpha)}$	0.95911 (P = 0.0103)		R^2(corrected)	0.1764
$\hat{\beta}$	0.000026759 (P=0.0669)		$\hat{S}_{MSY}$	17,368
$\hat{\sigma}_r^2$	0.3021		$\hat{U}_{MSY}$	0.46
$\hat{\sigma}_\varepsilon^2$	0.2613			

Spawning abundance that on average produces maximum sustained yield (S_{MSY}) was estimated by iteratively solving the following transcendental relationship:

$$1 = (1 - \hat{\beta}\hat{S}_{msy}) \exp(-\hat{\beta}\hat{S}_{msy}) \exp(\ln\hat{\alpha} + \hat{\sigma}_\varepsilon^2/2),$$

for $\hat{S}_{MSY}$ where $\hat{\sigma}_\varepsilon^2 = \hat{\sigma}_r^2 - \hat{\sigma}_{uv}^2 = 0.24$ and $\hat{\sigma}_r^2$ is the mean square error from the fitted model. The result is $\hat{S}_{MSY} = 17{,}368$ large chinook salmon (age 1.3 and older). An estimate of the optimal exploitation rate is $\hat{U}_{MSY} = \hat{\beta}\hat{S}_{MSY} = 0.46$.

While these estimates $\hat{S}_{MSY}$ and $\hat{U}_{MSY}$ have been adjusted for measurement error in the dependent variable, they have not been adjusted for measurement error in spawning abundance. In the log-linear transform of Cushing's model $\ln(R_y) = \ln(\alpha') - \beta'\ln(S_y) + \varepsilon_y$, estimates for parameters $\ln(\alpha')$ and β' are (see Section 1.6.1):

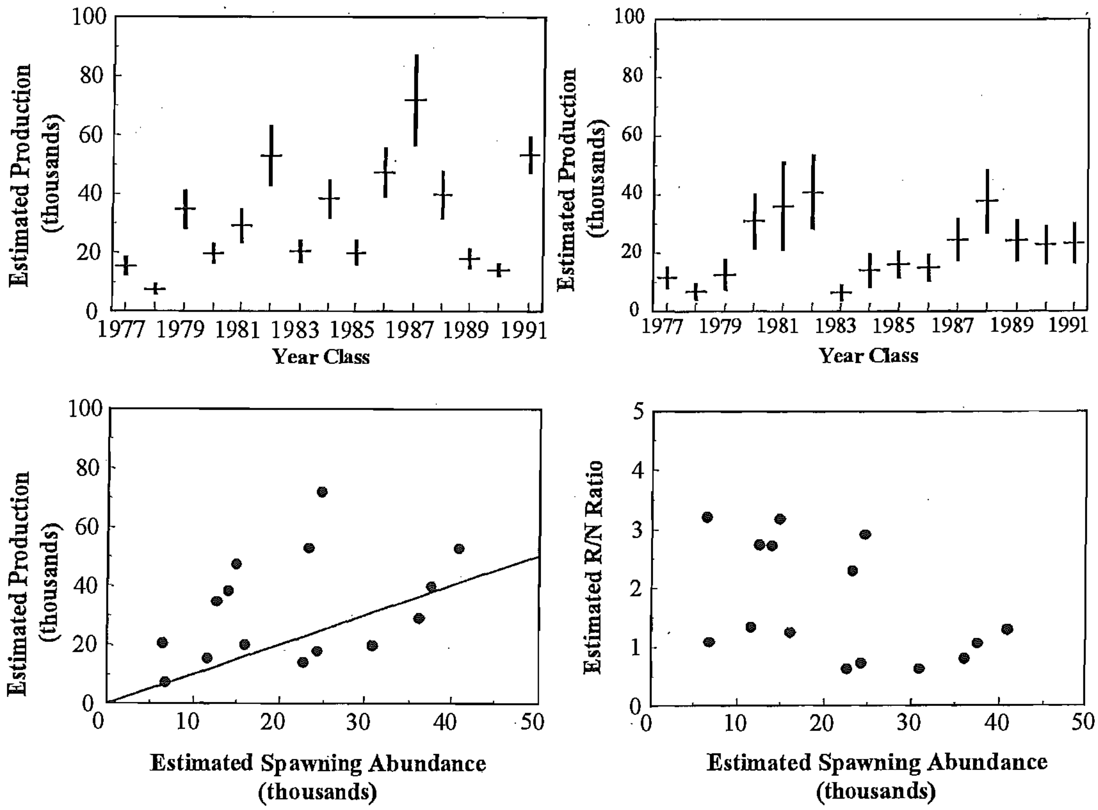


Figure 2.2. Estimated production $\hat{R}_y$ by year class, estimated spawning abundance $\hat{S}_y$ of salmon age 1.3 and older, and their estimated 95% confidence intervals.

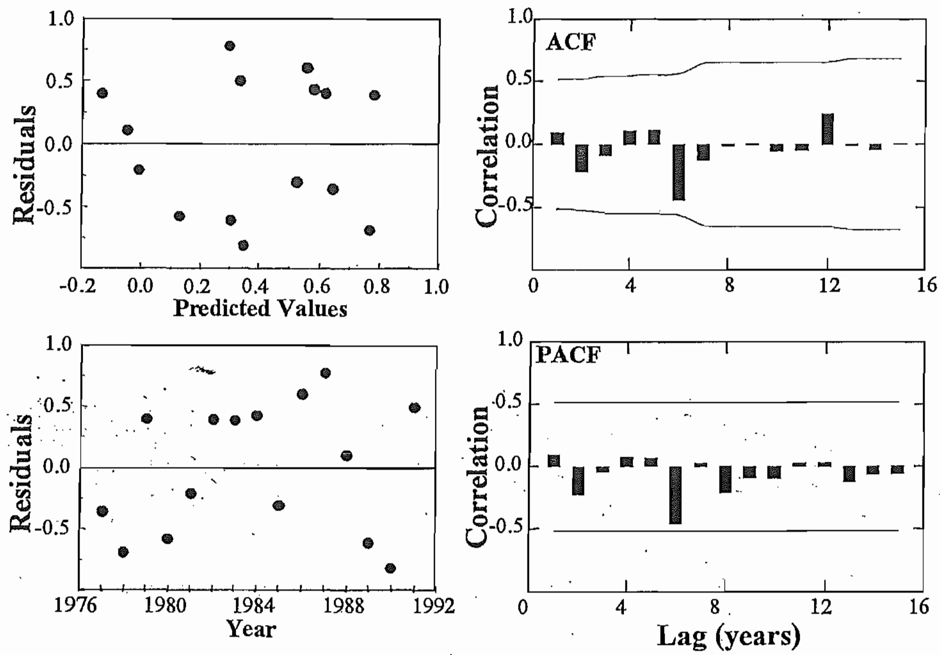


Figure 2.3. Residuals from the fit of Ricker's model plotted against predicted values of R_y and years (year classes) and autocorrelations (ACF) and partial autocorrelations (PACF) among residuals.

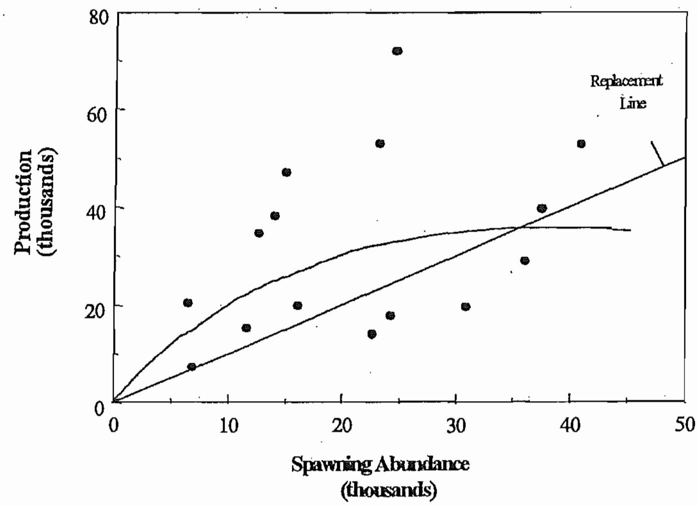


Figure 2.4. Estimated production of age 1.2-5 chinook salmon in year classes 1977-91 against the estimated spawning abundance of their parents age 1.3 and older for the population in the Stikine River. The curve represents production predicted with Ricker's model.

$$\hat{\beta}' = \frac{m_{XY}}{m_{XX} - \hat{\sigma}_u^2} = \frac{(n-1)^{-1}\sum_y^n(\ln\hat{S}_y - \overline{\ln\hat{S}})(\ln\hat{R}_y - \overline{\ln\hat{R}})}{(n-1)^{-1}\sum_y^n(\ln\hat{S}_y - \overline{\ln\hat{S}})^2 - \hat{\sigma}_u^2} = 0.5727$$

$$\ln(\hat{\alpha}') = \overline{\ln\hat{R}} - \hat{\beta}'\overline{\ln\hat{S}} = 4.5864$$

with a coefficient of determination (R^2) of 41%. Because there was little variation in the range of sampling variances for $\hat{R}_y$, weighted regression was not needed to stabilize conditional variances. The estimate for S_{MSY} with this model adjusted for measurement error in both dependent and independent variables is

$$\hat{S}_{MSY} = \sqrt[\hat{\beta}'-1]{(\hat{\alpha}'\hat{\beta}')^{-1}} = 17{,}730$$

where $\hat{\alpha}' = \exp(\ln\hat{\alpha}' + \hat{\sigma}_\varepsilon^2/2)$ and $\hat{\sigma}_\varepsilon^2 = \hat{\sigma}_r^2 - \hat{\sigma}_v^2 = 0.3124$. The estimate of the exploitation rate associated with S_{MSY} is:

$$\hat{U}_{MSY} = 1 - \hat{\beta}' = 0.43$$

The similarity in estimates of S_{MSY} (17,368 vs. 17,730) and estimates of U_{MSY} (0.46 vs. 0.43) from fitting both Ricker's and Cushing's models indicates that measurement error in spawning abundance representing 9% of overall variation in S_y was a negligible factor. Predicted values from both fits were similar over the range of data observed in this study, but diverged at higher numbers of spawners (Figure 2.5).

Simulation. The estimated 90% confidence intervals for $\hat{S}_{msy}$ and $\hat{U}_{msy}$ (Table 2.3) were calculated through bootstrap simulation of residuals in the fit of Ricker's model (see Section 1.8.2 for description of methods). One thousand new data sets were generated from the original by adding them to values predicted with Ricker's model, then Ricker's model was refit to each to produce new parameter estimates including $\tilde{\sigma}_\varepsilon^2$ as $\tilde{\sigma}_{\varepsilon(b)}^2 = \tilde{\sigma}_{r(b)}^2 - \hat{\sigma}_{uv}^2$. No adjustment was made for measurement error in spawning abundance. About 18% statistical bias was indicated in the estimate $\hat{S}_{MSY}$; virtually no statistical bias was indicated in the estimate $\hat{U}_{msy}$.

2.2.4. Discussion

Process error from environmental influences on survival rates dominates the analysis of MSY for the stock in the Stikine River. Measurement error was not a problem. Contrast in spawning abundance was good (6,424 to 40,488) with measurement error representing an estimated 9% of this contrast. Measurement error in estimated production was also negligible (CVs 10% on average). The fit of Ricker's log-transformed model is meaningful with the density-dependent parameter β being significantly larger than 0 with a 6 in 100 chance of a Type I error. Yet the fit of the model was poor (about 18% of variation explained) and the estimate of S_{MSY} carried some statistical bias (about 18%). Unexplained process error is the possible candidate for this poor result. Unfortunately, there is no direct information on marine survival rates for this stock, nor is there information from an indicator stock relevant to the Stikine River.

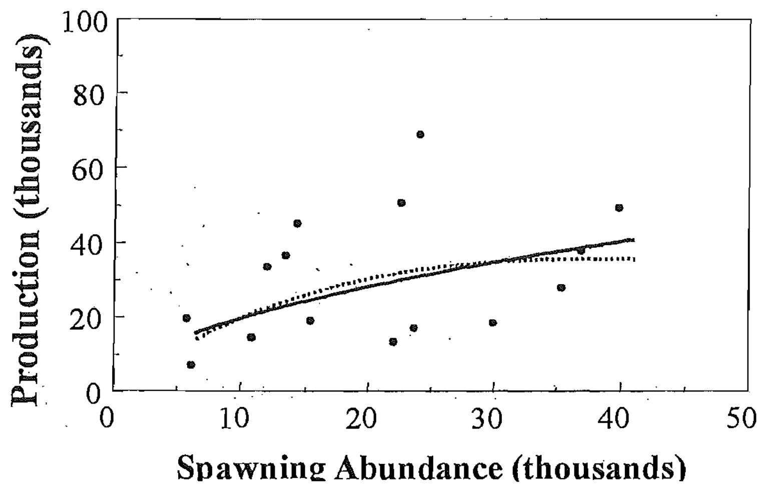


Figure 2.5. Estimates and predictions of production for the 1977-91 year classes of chinook salmon in the Stikine River. Predictions are from fits of Ricker's (dashed line) and Cushing's (solid line) models.

Table 2.3. Statistics from bootstrap simulations to estimate standard errors and 90% confidence intervals for S_{MSY} and U_{MSY}.

	S_{MSY}	U_{MSY}
Estimate	17,368	0.46
Bootstrap Mean	20,474	0.45
Standard Error	-	0.11
Lower Limit	11,838	0.26
Upper Limit	39,907	0.61

Re-instigation of a coded-wire-tagging program on the Stikine River has been recommended and supported with funds appropriated annually through the LOA among the states. Return of tags from catch sampling and from inriver sampling will be used to annually estimate harvest and estimate smolt abundance. Both statistics will be used to calculate marine survival rates, and these rates used as a covariate to "explain" much of the process error in future data on production.

The stock-production relationship for this stock in the Stikine River will be reanalyzed in three more years (year 2002). While there will be little new information on marine survival rates, more information will be available on variation in the expansion factors used to estimate spawning abundance. The expansion factor is currently based on three calculations on sampling conducted in two years. By 2002, there should be seven instances.

2.3. Taku River

2.3.1 Summary

Year Classes	S_{MSY} Range	$\hat{S}_{MSY}$	$\ln\hat{\alpha}$	$\hat{\beta}$	$\hat{U}_{MSY}$
1973 - 1994	30,000 – 55,000	35,938	1.4829	0.00001635	0.5875

An escapement goal range of 30,000 to 55,000 adult spawners (3-5 ocean-age) is recommended for this stock. This recommendation was accepted in the spring of 1999 by the Chinook Technical Committee, internal review committees of ADF&G and CDFO, and by the PSC's Transboundary Technical Committee. Statistics on harvest and spawning abundance by age group for 19 year classes (1973 to 1991) were used to estimate production along with estimates of smolt production for eight of these year classes plus estimates for the 1992 and 1993 year classes. The range was based on analysis of smolt production and growth occurring at extreme levels of spawning abundance. Evidence indicates that marine survival of smolts has been density-independent. Standard methods of spawner-recruit analysis based on regressing production against spawning abundance were tried, but rejected as producing unreliable results due to large measurement error in estimates of spawning abundance. Parameter estimates for Ricker's model for use in the CTC model were based on the escapement goal range and estimated adult-to-smolt ratios, not on regression.

Primary Reference: McPherson, S. A. and D. R. Bernard. *In Prep*. Optimal production of chinook salmon from the Taku River. Alaska Department of Fish and Game, Sport Fish Division, Fishery Manuscript No.99-__, Anchorage.

Primary Contact: Scott McPherson, ADF&G (907-465-4313; Scott_McPherson@fishgame.state.ak.us)

2.3.2. Stock Description and Stock Assessment Data

Watershed Description: The Taku River is a large watershed with its origin in British Columbia and its terminus in SEAK (Figure 2.1). The drainage covers approximately 17,094 km^2 and discharge just below the international border ranges from a winter low of 60 m^3/sec to 1,097 m^3/sec in June, on average. The mainstem Taku River is turbid from late spring through early fall. Most chinook salmon spawning occurs in clearwater tributaries to the Taku River, such as the Nakina, King Salmon, Kowatua, Hackett, and Nahlin Rivers. The drainage is pristine--little development has occurred in the drainage; no roads connect the drainage to any road system, miniscule logging has occurred, no dams have been constructed and one mineral mine presently operates in the lower Canadian portion of the drainage.

Stock Description: The Taku River chinook salmon stock is a "spring" stock. Mature fish migrate through a marine terminal sport fishery located near Juneau, Alaska, from late April through mid-July. These fish enter the lower Taku River drainage from early May through late

July and migrate upriver to Canada to a variety of spawning streams. Spawning activity takes place from late July to mid-September in year *y*. Taku River chinook salmon produce primarily yearling smolt. Fry rear primarily in the main river as well as in some larger tributaries through the following spring, before migrating to sea in year *y*+2. After entering ocean waters, young rear offshore out of reach of fisheries. These fish mature and migrate back through SEAK after one to five years at sea. Presently, the Taku River chinook run is targeted in a marine sport fishery centered near Juneau, Alaska. Incidental harvests are taken in the experimental openings in the commercial troll fishery in May and June. The tail end of the spawning migration is also subject to incidental capture in a commercial marine gillnet fishery in Taku Inlet that targets sockeye salmon and begins in late June. Small catches of Taku chinook occur in minor commercial troll openings in May and June. Small harvests are taken in inriver commercial and sport fisheries in the Canadian portion of the Taku River.

Hatchery Influence: Hatchery chinook salmon have not been released in this transboundary river nor in nearby waters. Strays from hatcheries have not been found on the spawning grounds and hence, hatchery influence to this stock is considered nil. This is judged from several thousand spawners that are sampled annually for biological data, including recovery of coded-wire-tags (CWTs).

Escapement Estimation Method: Chinook salmon escapements in the Taku River have been assessed with helicopter surveys conducted annually since 1973 (Kissner and Hubartt 1986). Chinook salmon larger than approximately 660 mm (mid-eye to fork of tail) were included in counts. These are typically 3-ocean age and older fish. Smaller fish were not counted because they could not be distinguished from other species. Since counts were highly correlated across tributaries, they were summed to represent an index count of the entire population of large chinook salmon in the drainage.

Aerial counts have been expanded to estimate total abundance based on five years (1989, 1990, and 1995-1997) when both helicopter surveys and mark-recapture estimates of large spawners were available (Table 2.1). Details concerning expansions and sampling errors associated with these expansions can be found in McPherson and Bernard (*In Prep.*).

Age and Sex Composition of Escapement: Fish maturing at a young age (age 1.1 and 1.2) are almost exclusively males, while older fish (age-1.3, 1.4, and 1.5) are 50% females, on average. Ages 1.2, 1.3, and 1.4 dominate the annual spawning population. Details concerning age composition and sample sizes associated with annual estimated age compositions of the Taku River stock of chinook salmon can be found in McPherson and Bernard (*In Prep.*). Samples of 1,000 or more have been taken annually from the escapements since 1973 by Alaskan and/or Canadian cooperators.

Escapement and Production: Because only mature Taku River chinook salmon (ages 1.2, 1.3, 1.4, and 1.5) are believed to be vulnerable to fisheries, production estimates are based on terminal returns. No adjustment was made for harvest of immature fish or younger mature fish (calculating adult equivalents). Spawning abundance by age and estimated annual harvests in terminal fisheries by age were used to develop estimates of total recruits from spawning populations in the years 1973–1991. Table 2.4 provides annual estimates of spawning abundance and estimated resultant production from these escapements along with standard errors of estimates (see McPherson and Bernard *In Prep.* for details concerning estimation methodology).

Smolt Production: Stock assessment has included a tagging program to estimate abundance of smolts. Smolts and/or fingerlings were implanted with coded-wire tags from the 1975–1981 broods (year classes) and from the 1991–1995 broods. Young fish were captured in the lower river near or downstream of the international border with baited minnow traps and also with rotary screw traps in some later years. The fraction of year class y tagged in year $y+2$ as smolts was estimated by summing data on adults of that year class sampled on the spawning grounds or caught at Canyon Island (tagging site 2 km below international border) in years $y+3$, $y+4$, $y+5$, and $y+6$. Recovery of coded-wire tags from adults on the spawning grounds showed that tagged smolts represented all subpopulations in the Taku River in near equal proportion. The number of tagged smolt in year $y+2$ was divided by the estimated marked fraction of adults of year class y to estimate the number of smolt emigrating that year as per a simple, two-event mark-recapture experiment on a closed population (Seber 1982:60). Because too few smolt were recaptured for some year classes, estimates of smolt abundance are available only for year classes 1975, 1976, 1979, and 1991–1995. Table 2.5 has the estimates of female spawners, smolt production, and adult production for these year classes.

2.3.3. Stock-Production Analysis – Stock and Production Data

Measurement Error: Because values of R_y and S_y are unknown for the Taku River population, their estimates were used in the analysis as substitutes. Use of estimates introduced measurement error into both independent and dependent variables. As per Section 1.5.2 log-normal measurement error can itself be estimated when sampling variances are calculated. For measurement error in spawning abundance:

$$V[\ln(\hat{S})] = V[\ln(S)] + \sigma_u^2 .$$

These variances are unknown, but can be estimated as $v[\ln(\hat{S})]$ and $\hat{\sigma}_u^2$ such that:

$$v[\ln(\hat{S})] = \frac{\sum[\ln(\hat{S}_y) - \overline{\ln(\hat{S})}]^2}{n-1} = 0.3033.$$

Table 2.4. Estimated spawning abundance of females $\hat{S}_{y(f)}$ and estimated production $\hat{R}_y$ of large adults along with their standard errors (*se*) and coefficients of variation (*cv*) for the 1977–1991 year classes of chinook salmon in the Taku River and estimates of spawning abundance for the 1992-1997 year classes.

Year Class	$\hat{S}_{y(f)}$	$se(\hat{S}_{y(f)})$	$cv(\hat{S}_{y(f)})$	$\hat{R}_y$	$se(\hat{R}_y)$	$cv(\hat{R}_y)$
1973	8,929	3,864	43.3%	19,931	5,266	26.4%
1974	9,824	4,236	43.1%	75,456	22,913	30.4%
1975	4,593	2,139	46.6%	87,450	23,384	26.7%
1976	15,165	6,478	42.7%	65,457	16,615	25.4%
1977	20,466	8,678	42.4%	34,312	11,164	32.5%
1978	9,143	3,997	43.7%	16,547	4,828	29.2%
1979	10,997	4,991	45.4%	39,833	9,288	23.3%
1980	21,228	9,450	44.5%	58,388	14,691	25.2%
1981	25,024	11,144	44.5%	45,833	12,442	27.1%
1982	12,396	5,426	43.8%	60,035	15,423	25.7%
1983	4,120	1,903	46.2%	37,079	8,341	22.5%
1984	10,091	4,720	46.8%	85,187	13,764	16.2%
1985	17,447	7,820	44.8%	62,650	11,097	17.7%
1986	21,700	9,523	43.9%	61,805	14,530	23.5%
1987	12,607	5,778	45.8%	95,777	23,601	24.6%
1988	21,864	9,742	44.6%	80,004	21,182	26.5%
1989	17,580	4,827	27.5%	67,788	14,651	21.6%
1990	26,749	5,831	21.8%	34,078	4,194	12.3%
1991	27,435	11,842	43.2%	196,114	14,153	7.2%
1992	55,889	22,902				
1993	66,125	27,097				
1994	48,368	19,820				
1995	35,162	5,060				
1996	81,416	9,048				
1997	114,828	17,888				

Table 2.5. Estimated abundance of females, smolts, production, and estimated mean fork lengths for smolts for several year classes of chinook salmon in the Taku River. Standard errors are in parentheses. Standard errors for ratios were approximated by the delta method (Seber 1982).

Year Class	Females	Smolts	Mean length fl (mm)	Smolts per Female	Production	Adults per Smolt
1975	4,593 (2,139)	1,189,118 (174,197)	79	258.9 (126)	87,450 (23,384)	0.074 (0.0224)
1976	15,165 (6,478)	1,549,052 (374,227)	71	102.1 (50)	65,457 (16,615)	0.042 (0.0148)
1979	10,997 (4,991)	661,150 (97,648)	74	60.1 (29)	39,833 (9,288)	0.060 (0.0166)
1991	27,435 (11,842)	2,098,862 (295,390)	80	76.5 (35)	196,114 (14,153)	0.093 (0.0148)
1992	22,935 (10,391)	1,968,167 (438,569)	73	85.8 (43)	79,307[a]	0.0403
1993	29,976 (13,573)	1,267,907 (564,432)	78	42.3 (27)	19,114[b]	0.0151
1994	31,553 (13,565)	1,328,553 (352,068)	76	42.1 (21)		—
1995	18,942 (2,891)	1,898,233 (626,335)	77	100.2 (36)		—

[a] Estimate is based on final estimate of spawning abundance and preliminary statistics on catch.
[b] Estimate is based on imputing production of age 1.4 and 1.5 salmon as the average (34% of production) over all age groups for 1973-91 year classes.

$$\hat{\sigma}_u^2 = \frac{\sum \hat{\sigma}_{u,y}^2}{n} = \frac{cv^2(\hat{S}_y)}{n} = 0.1832$$

Note these calculations show estimated measurement error comprised 60% (=0.1832/0.3033) of all variation in estimated spawning abundance.

Log-normal measurement error in estimates of production was estimated as (see Section 1.5.2):

$$\hat{\sigma}_v^2 = \frac{\sum \hat{\sigma}_{v,y}^2}{n}$$

$$\hat{\sigma}_{v,y}^2 = v[\ln(\hat{R}_y)] \cong v(\hat{R}_y)\hat{R}_y^{-2} = cv^2(\hat{R}_y).$$

For the population in the Taku River, $\hat{\sigma}_v^2 = 0.0583$. Estimated measurement error for the estimated log of the production-to-spawner ratio $\hat{R}_y/\hat{S}_y$ is $\hat{\sigma}_{uv,y}^2 = cv^2(\hat{R}_y) + cv^2(\hat{S}_y)$. The average over all year classes is $\hat{\sigma}_{uv}^2 = 0.2415$.

The magnitude of measurement error in estimates of production and spawning abundance for this stock was graphically displayed with the aid of simulation (Figure 2.5). A log-standard normal variate was randomly selected for each estimate of harvest, spawning abundance, and relative age composition, then transformed into a variate with the appropriate mean and variance, thereby creating a new set of statistics from the original. These new simulated statistics were multiplied and their products added appropriately to obtain a simulated set of data pairs $\{\tilde{R}_y, \tilde{S}_y\}$. The process was used to create 950 pairs. The cloud of simulated points spreads out horizontally from close on the y-axis out to about twice the highest estimate of spawning abundance. The cloud also spreads vertically topped by a curious wisp of points set above the cloud. The wisp is a result of the relatively precise estimate of production for the 1991 year class [$cv(\hat{R}_{91}) = 7.0\%$] (due to mark-recapture experiments) and poor precision in estimates of the females that spawned them [$cv(\hat{S}_{91}) = 43\%$] from the expansion of an aerial survey. This contrast in precision laterally flattens and elongates the cloud of simulated points for this year class. Simulated points for the 1991 year class are set above the rest because their production was atypically strong.

Parameter Estimates: Two models were used in the analysis: Ricker's exponential function $R_y = \alpha S_y \exp(-\beta S_y)\exp(\varepsilon_y)$ and Cushing's power function $R_y = \alpha' S_y^{\beta'} \exp(\varepsilon_y)$. The latter is an approximation to the former that allows incorporation of measurement error in spawning abundance in the analysis. The term ε_y represents process error in both models where $\varepsilon_y \sim N(0, \sigma_\varepsilon^2)$. Parameters were estimated for the linear form of Ricker's model $\ln(R_y) - \ln(S_y) = \ln(\alpha) - \beta S_y + \varepsilon_y$ (Table 2.6) with the computer program Systat™. Because estimated precision for brood years 1989-91 was considerably improved over earlier year classes, parameters were estimated with unweighted regression and with regression where the dependent variable was weighted by $1/\hat{\sigma}^2_{uv,y}$. No autocorrelation among residuals or higher order influence of spawning abundance could be found (as per methods in sections 1.6 and 1.5.1.2). Predictions by the fitted, untransformed model and the original data are given in Figure 2.6. Spawning abundance that on average produces maximum sustained yield (S_{MSY}) was estimated by iteratively solving the following transcendental relationship:

$$1 = (1 - \hat{\beta}\hat{S}_{msy})\exp(-\hat{\beta}\hat{S}_{msy})(1+\tau)^{-1}\exp(\ln\hat{\alpha} + \hat{\sigma}_\varepsilon^2/2)$$

for $\hat{S}_{MSY}$ where $\hat{\sigma}_\varepsilon^2 = \hat{\sigma}_r^2 - \hat{\sigma}_{uv}^2 = 0.12$ for both unweighted and weighted regressions, $\hat{\sigma}_r^2$ is the mean square error from the fitted regression, and the male-to-female ratio $\tau = 1$. Little difference was seen between statistics for the unweighted and weighted regression ($\hat{S}_{MSY} = 11{,}629$ vs. 10,416 females and $\hat{U}_{MSY} = \hat{\beta}\hat{S}_{MSY} = 62\%$ vs. 64%) (Table 2.6). These estimates have been adjusted for measurement error in the dependent variable, but not for measurement error in the independent variable.

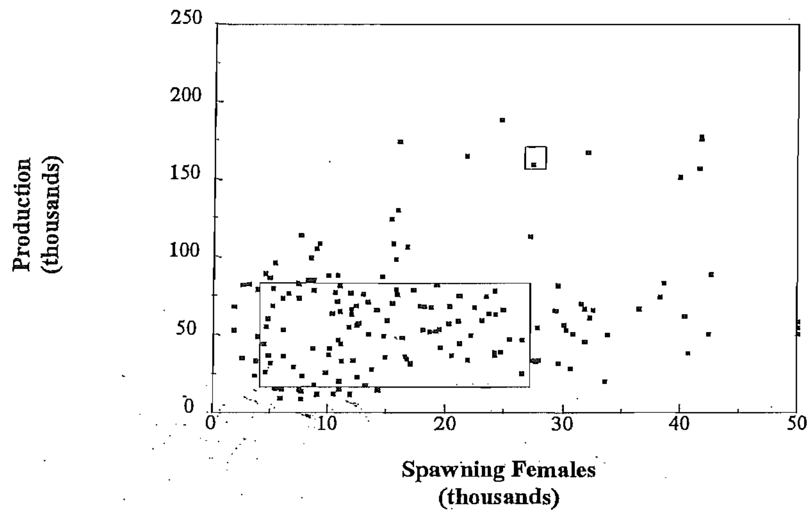


Figure 2.6. Spawning abundance of females and associated production simulated from the measurement error in original data. Boxes correspond to the range of original data. Bands along the right edge of the plot correspond to a few instances with spawning abundance beyond 50 thousand females.

Adjustment for measurement error from estimating spawning abundance dramatically changes perspectives on values of S_{MSY} and U_{MSY}. In the log-linear transform of Cushing's model $\ln(R_y) = \ln(\alpha') - \beta' \ln(S_y) + \varepsilon_y$, estimates for parameters $\ln(\alpha')$ and β' are (see Section 1.6.1)

Table 2.6. Estimated parameters for unweighted and weighted regression on the log-linear transform of Ricker's model on estimates of production and spawning abundance of chinook salmon in the Taku River.

	Unweighted	Weighted
$\ln(\hat{\alpha})$	2.2240 (P < 0.0001)	2.3191 (P < 0.0001)
$\hat{\beta}$	-0. 00005338 (P=0.0146)	-0. 00006184 (P=0.0035)
R^2(corrected)	0. 2621	0. 3673
$\hat{\sigma}_r^2$	0. 3606	0. 3646
$\hat{\sigma}_\varepsilon^2$	0.1191	0.1231
$\hat{S}_{MSY}$ (females)	11,629	10,459
$\hat{U}_{MSY}$	0.62	0.65

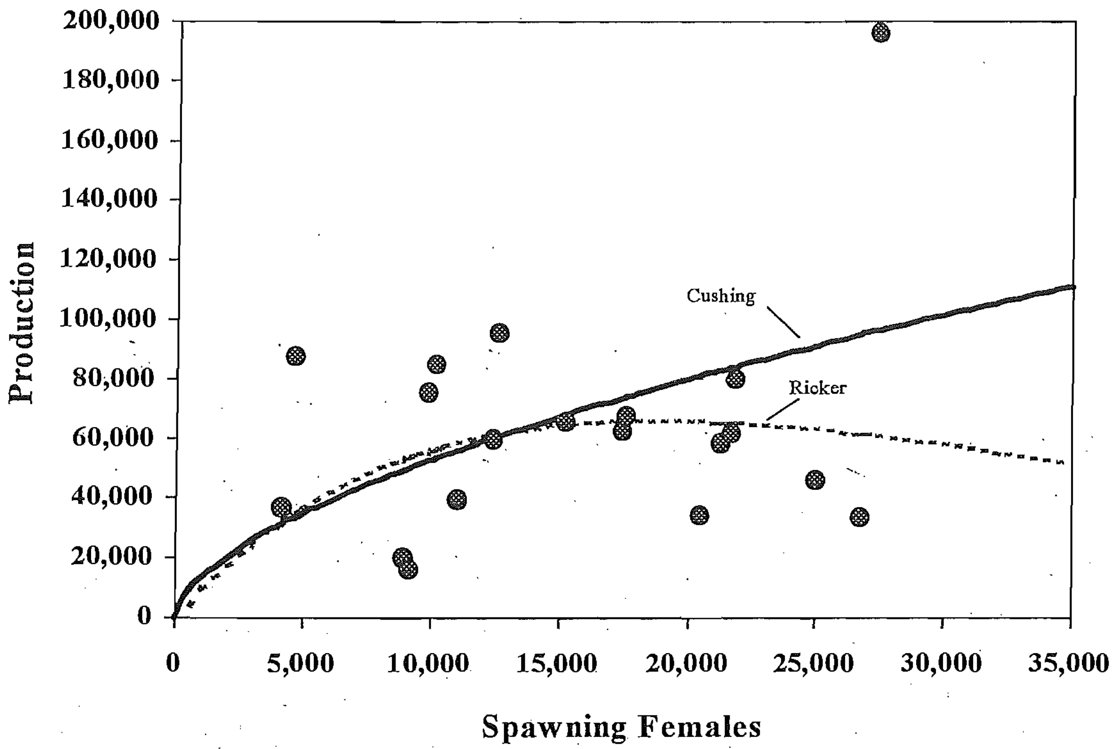


Figure 2.7. Estimated production of age 1.2-5 chinook salmon in year classes 1973–1991 against the estimated abundance of females that spawned them along with curves corresponding to least-squares fits of Ricker and Cushing models to all data.

$$\hat{\beta}' = \frac{m_{XY}}{m_{XX} - \hat{\sigma}_u^2} = \frac{(n-1)^{-1}\sum_y^n(\ln\hat{S}_y - \overline{\ln\hat{S}})(\ln\hat{R}_y - \overline{\ln\hat{R}})}{(n-1)^{-1}\sum_y^n(\ln\hat{S}_y - \overline{\ln\hat{S}})^2 - \hat{\sigma}_u^2} = 0.5926$$

$$\ln(\hat{\alpha}') = \overline{\ln\hat{R}} - \hat{\beta}'\overline{\ln\hat{S}} = 5.2726$$

The estimate for S_{MSY}(females) with this model adjusted for measurement error in both dependent and independent variables is:

$$\hat{S}_{MSY} = \sqrt[\hat{\beta}'-1]{\frac{1+\tau}{\hat{\alpha}'\hat{\beta}'}} = 30,917,$$

Table 2.7. Estimated parameters for the log-linear transform of Cushing's power function fit to all data on production and spawning abundance of chinook salmon in the Taku River and fit to data with the 1991 year class excluded.

	All Data	1991 Excluded
$\ln(\hat{\alpha}')$	5.2726	8.9688
$\hat{\beta}'$	0.5926	0.1983
R^2(corrected)	0.2841	0.0389
$\hat{\sigma}_r^2$	0.3706	0.2666
$\hat{\sigma}_\varepsilon^2$	0.3123	0.2083
$\hat{S}_{MSY}$	30,917	4,602
$\hat{U}_{MSY}$	0.41	0.80

where $\hat{\alpha}' = \exp(\ln\hat{\alpha}' + \hat{\sigma}_\varepsilon^2/2)$ and $\hat{\sigma}_\varepsilon^2 = \hat{\sigma}_r^2 - \hat{\sigma}_v^2 = 0.29$. The estimate of the exploitation rate associated with S_{MSY} is:

$$\hat{U}_{MSY} = 1 - \hat{\beta}' = 0.41.$$

These estimates of S_{MSY} and U_{MSY} have been adjusted for measurement error in both dependent and independent variables. Parameter estimates are reported in Table 2.7, and predicted values in Figure 2.6.

Inspection of Figure 2.6 shows that the 1991 year class is unusually influential in determining the fit of Cushing's model to data from the Taku River. If information on that year class is excluded from the data and Cushing's model refit, the resulting statistics differ dramatically (Table 2.7) with $\hat{S}_{MSY}$ dropping to 4,602 and $\hat{U}_{MSY}$ rising to 80%.

3.3.4. Stock-Production Analysis - Smolt Data

Evidence for Density-dependence: An analysis of the more precise statistics on production and on the data in Table 2.5 on smolt production reveals evidence to support the following:

- a wide range of spawning abundance over the years, with an even wider range to come;
- density-dependent survival in the early freshwater life of young chinook salmon;
- potential density-independent survival in the later freshwater life of young chinook salmon;
- density-independent survival of smolts at sea; and
- an upper bound on the production of smolts from the Taku River.

The range in spawning abundance observed over the years in not an artifact of measurement error. Based on expansions of counts from aerial surveys, the lowest spawning abundance of females were 4,120 in 1983 and 4,593 in 1975 (Table 2.4). In 1990, the spawning abundance was estimated at 26,749 females with a mark-recapture experiment. While relative precision on the lower estimates (1983 and 1985) is not good, that on the higher estimate (1990) is good enough to show with high probability ($P<0.05$) that the six-fold increase it represents is real. The same is true for spawning abundance estimated in 1997, 2.6 times higher than in 1990 and 17 times higher than in 1983.

Density-dependent survival of young is indicated because the highest relative production of adults occurred when spawning abundance was lowest in 1975 and 1983 (Table 2.8). The probability from random chance alone (measurement error) that the two smallest numbers of spawners would have the highest relative rate of production over 19 year classes is 0.0058 $[=2(\frac{1}{19})(\frac{1}{18})]$. The next estimated highest ratio was 8.442 for the 1984 year class spawned by an estimated 10,091 females; the lowest estimated ratio was 1.274 for the 1990 year class.

The limited range in size of smolt (Table 2.5) is evidence that this density-dependent survival has an early influence on young chinook, at least over the years for which we have estimates. The range in estimated smolt abundance (661,150 to 2,098,862) is statistically significant ($P<0.01$) while the corresponding sizes of these smolts were similar (74 mm vs. 73 mm FL). Lack of density-dependent growth is evidence that rearing habitat was not a compensatory limiting factor, nor was predation a detectable depensatory factor in mortality of young. The range of 71 to 80 mm FL for smolts expressed in Table 2.5 is representative of all studied year classes.

Table 2.8. Extremes in estimates of female spawning abundance and their ratios with subsequent production. Standard errors are in parentheses with SE for ratios approximated with the delta method (Seber 1982:7-9)

Year Class	$\hat{S}_{y(f)}$		$\hat{R}_y/\hat{S}_{y(f)}$	
1975	4,593	(2,139)	19.040	(10.225)
1983	4,120	(1,903)	9.000	(4.624)
1990	26,749	(5,831)	1.274	(0.319)
1997	70,429	(11,039)		

Comparison of the estimated number of adults produced from an estimated number of smolts points to density-independent marine survival. Estimated smolt abundance from the 1976 and 1991 year classes (1.55 vs. 2.10 million, Table 2.5) was not significantly different ($P>0.20$); an estimated 4.2% returned as adults for the earlier year class and 9.3% for the later year class. While the estimated numbers of smolt are not statistically different, the return rates are ($P<0.01$). Estimated size of smolts for these year classes (71 and 80 mm FL) do cover the observed range. In contrast, estimated smolt abundance was significantly different for the 1975 and 1979 year classes (1.20 vs. 0.66 million, $P<0.01$) while their return rates were similar (0.074 vs. 0.060; $P>0.50$). Estimated smolt size for the 1975 and 1979 year classes are closer: 74 mm vs. 79 mm FL.

The evidence in the smolt information underpinning a ceiling on the number of smolts produced each year by the Taku River is circumstantial. Early density-dependence in the freshwater existence of chinook salmon is the result of limited, high quality spawning habitat or limited rearing habitat for emerging young. If the earliest determinants of year-class strength are the only density-dependent factors in the life history of the population (as may be indicated), the highest production observed from a given spawning abundance captures the most information on density dependence.[1] For these reasons, the highest production ratio (smolts/female) for a given spawning abundance (females) is the best reflection of the effect of density-dependent survival on young. The year classes with the highest production ratios are 1975, 1976, 1991, 1992, and 1995 (Table 2.5). The other year classes (1979, 1993, and 1994) had estimated smolt production unexpectedly low given the estimated abundance of female parents. If there is a ceiling on smolt production in the Taku River, smolt production should follow an asymptotic, density- dependent relationship (Figure 2.7). However, smolt production is too similar among the year classes and the precision in estimates too poor to distinguish a true asymptote. Under these circumstances, the average of the four highest smolt estimates (1976, 1991, 1992, and 1995), 1.879 million smolt with a SE of 0.451 million, is a minimal estimate of the ceiling.

MSY Escapement Level: The most defensible estimate for S_{MSY} from the auxiliary analysis lies in the range of 30,000 to 55,000 large spawners. This range was chosen as twice the number of females that had produced near or at the maximum number of smolts, i.e., the four highest estimates of smolt production in Figure 2.7. In 1976 an estimated 15,165 large females spawned and produced an estimated 1.55 million smolts that went to sea in 1978; in 1991 an estimated 27,435 large females spawned and produced an estimated 2.1 million smolts that went to sea in 1993 (Table 2.5). Given density-independent marine survival, maximizing smolt production with a minimal number of spawners will result on average in production near to MSY, which is the lower end of the range.

The upper end of the S_{MSY} range reflects the higher levels of spawning abundance where high smolt production was maintained. Smolt estimates for the 1993 and 1994 year classes are significantly different from the higher smolt estimate from the 1991 year class. These may or may not be the result of density-independent factors, but they represented no improvement in smolt production beyond the chosen level.

Lower levels of spawning abundance were considered too risky when survival in freshwater has a significant density-independent component. The 1975 year class began with fewer females (an estimated 4,593) than in 1976 and produced marginally fewer smolt (1.2 million). Both these statistics are precise enough to feel confident of the reality of low spawning abundance and subsequently good smolt production. However, the 1979 year class produced only 0.7 million smolt from a similar spawning abundance. This difference in smolt production of 0.5 million is statistically significant ($P<0.01$) and represents density-independent freshwater survival.

[1] For instance, if 27,345 females in 1991 produced 2.1 million smolts (Table 2.5), and if about the same number of females three years later (31,553) produced an estimated 1.3 million smolts, the difference of 0.8 million smolts is significant ($P <0.10$), and represents the effects of density-independent factors alone.

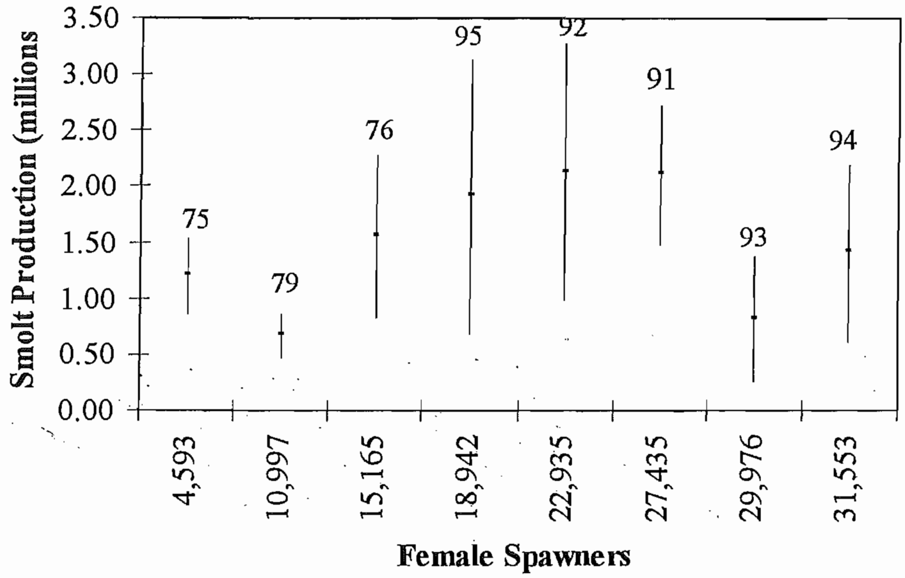


Figure 2.8. Estimated smolt production and estimated abundance of female parents for the 1975, 1976, 1979, and 1991–1995 year classes. Intervals on smolt production are approximate 95% confidence intervals.

A point estimate for S_{MSY} of 35,938 can be derived from the range of 30,000 to 55,000 large spawners (age 1.3 and older). Eggers (1993) showed that for sockeye salmon (*O. nerka*), a range of 80% to 160% of S_{MSY} produces a yield that is ≥80% of MSY. If this same relationship is true for chinook salmon, two estimates of S_{MSY} for the chinook salmon in the Taku River would be 30,000/0.80 = 37,500 and 55,000/1.6 = 34,375. The average of these two estimates would be approximately 36,000 large spawners.

Parameter Estimates: Estimates of α and β (Ricker's model) for use in the CTC model were also estimated from smolt data. Smolt production over year classes with spawning escapements from 30,000 to 55,000 (1976, 1991, 1992, 1995) averaged 1,878,579 smolts (Table 2.5). The geometric mean of estimated adult-to-smolt ratios is 0.0468 (Table 2.5) making the expected production at S_{MSY} to be 87,982 [=(0.047)(1,878,579)] salmon. Given Hilborn's approximation (see equation below), there are two equations with two unknowns:

$$87{,}892 = \alpha(35{,}938)\exp[-\beta(35{,}938)] \text{ :Ricker's model}$$

$$35{,}938 = \frac{\ln(\alpha)}{\beta}(0.50 - 0.07\ln\alpha) \text{ :Hilborn's approximation}$$

Solving these equations produces estimates $\hat{\alpha} = 4.4055$ and $\hat{\beta} = 0.00001635$. The estimate for the exploitation rate associated with S_{MSY} is 0.59 $(=\hat{U}_{MSY})$.

2.3.5. Discussion

The recommendation of 30,000 to 55,000 large spawners for a range around S_{MSY} derived from the smolt data, though somewhat subjective, is more defensible than estimates from more orthodox methods. The contrast among estimates of S_{MSY} and U_{MSY} for the stock in the Taku River exposes the danger inherent in ignoring large measurement error in estimates of spawning abundance. The reason for this danger can be seen in Figure 2.5; measurement error artificially elongates the data cluster thereby artificially increasing the estimate of α as noted by Hilborn and Walters (1992:288) when they wrote: "... *in the extreme* (R_y) *may appear to be independent of* (S_y) .... *(leading) to overestimates of the slope of the recruitment curve for low spawning stocks.*" While correcting the data for measurement error with the adaptation of Cushing's model probably produced more reasonable estimates of S_{MSY} and U_{MSY}, the influence of the 1991 year class was critical to these estimates. Since production in this year class was largely due to unusually favorable, density-independent marine survival rates, estimates of S_{MSY} and U_{MSY} from using Cushing's model are also suspect. In short, estimates of spawning abundance in the Taku River probably carry too much measurement error (an estimate 60% of all variation in estimated spawning abundance) to be effectively negate their its influence on regression analysis to estimate S_{MSY}.

Stability of environment, at least around average conditions, is presumed under traditional statistical analysis of stock-recruit data; the same is true under our scientific analysis of smolt production for chinook salmon of the Taku River. Evidence in our data for such stability is:

- Smolt sizes were essentially the same for early and late year classes in the series;
- Maximum production of smolt is similar across year classes with about 15,000 to 27,500 female spawners; and
- There was negligible or no loss of habitat during our series from land development, land use, or human habitation.

Evidence in our data against such stability of environment can be found in the marine survival for the 1991 year class (0.093) which was 63% higher ($P<0.01$) than the average for year classes two decades earlier (0.057). However, the 1991 year class is an outlier and other year classes from the 1990s are not returning at the same rate (Table 2.5).

Better precision in future statistics would improve the dependability of more traditional stock-recruit analyses. Estimates of smolt abundance, especially for the 1997 year class, may provide strong clues as to how many smolts the Taku River can produce. If smolt production from this exceptionally large number of spawners is considerably above the 2.1 million maximum estimated from data available now, the current estimate of S_{MSY} would have to be increased. Unfortunately, investigating productivity of chinook salmon is not for the impatient. Smolt abundance for the 1997 year class will not be available until 2001 or 2002. Complete information on production of that year class will be available a year or two later. For these reasons, plans are to reanalyze data to more definitively estimate S_{MSY} in the year 2004.

3. COLUMBIA RIVER

3.1. Introduction

Annual CTC escapement assessments include five indicator stocks of chinook salmon that originate in the Columbia River watershed. In addition to these analyses, the CTC has incorporated ten escapement goals for the Columbia River watershed into the chinook model; three of these goals are for wild stocks while seven are for hatchery stocks. A summary of goals used in the past by the CTC for the Columbia River watershed is provided in Table 3-1.

After an earlier review of information associated with escapement goals coastwide, the CTC judged that none of the stocks of chinook salmon from the Columbia River had biologically based escapement goals (CTC 1998). Since that review, the CTC has collected information on escapement and production for several wild stocks in the Columbia River Basin, including fall bright stocks in the Lewis River, the Deschutes River, the Snake River, and the Hanford Reach (Bernard and Clark 1999). With the exception of the fall migrating stock in the Lewis River, the CTC was still unable to develop maximum sustained yield or other biologically based escapement goals for bright fall chinook salmon returning to the Columbia River watershed because of a variety of technical issues concerning the data. To resolve these issues for the various stocks of bright fall chinook, the USCTC has supported the funding of research by the Washington Department of Fish and Wildlife (WDFW 1999) to identify biologically based goals for the Columbia River bright stocks of upriver fall chinook salmon. The final report for this study is scheduled for late fall of 1999. It is envisioned that the CTC will be able to make determinations of appropriate goals for these stocks based upon this planned research in the spring of 2000.

Due to a lack of detailed stock, production, and associated data for other Columbia River stocks of chinook salmon, the CTC has had difficulty confirming or improving and replacing other existing escapement goals for Columbia River chinook salmon stocks. As a result, the majority of Columbia River chinook salmon stocks remain in the list of coastwide stocks for which an agreed to biological escapement goal has yet to be identified. However, an agreed to interim goal was calculated for the component of the Columbia upriver summer escapement indicator stock (which is comprised of both Upper Columbia and Snake River summer chinook passing Bonneville Dam) which passes upstream of Rock Island Dam. The agreed to interim goal is based upon CTC model data initially developed to complete retrospective analyses associated with the treaty annex negotiations (CTC 1998). The analysis producing the interim goal is documented in this report and will be used by the CTC until a more thorough analysis based upon detailed stock-production information is completed. In order to ensure the timely completion of a thorough analysis, the USCTC has supported the funding of research by the Washington Department of Fish and Wildlife to develop a detailed stock-production data base and identify a biologically based goal for the Columbia River summer stock of chinook salmon. The final report for this study is scheduled for fall of 2000.

The remainder of this chapter provides a description of progress to date in estimating escapement goals associated with maximum sustained yield for the Lewis River fall and the Columbia upriver summer runs of chinook salmon.

Table 3.1. Stocks of chinook salmon in the Columbia River and their escapement goals as defined and used by the CTC.

Indicator Stock	Wild or Hatchery Stock	Escapement Indicator Stock	Goal Used in Past CTC Analyses to Assess Rebuilding	CTC Model Stock	Goal Used in Past CTC Chinook Model Analyses
Columbia Upriver Spring	wild	yes	84,000 natural spawners above Bonneville Dam	no	–
Columbia Upriver Summer	wild	yes	85,000 adults above Bonneville Dam	yes	35,600 Upper Columbia adult spawners
Columbia Upriver Bright	wild	yes	40,000 natural adult spawners above McNary Dam	yes	40,000 natural adult spawners above McNary Dam
Lewis River Fall	wild	yes	5,700 adult fall chinook in the Lewis River	yes	5,700 adults in the Lewis River
Deschutes R.	wild	yes	No Goal	no	–
Snake R Fall	wild	no	-	yes	3,430
Columbia Midriver Bright	Hatchery	no	-	yes	12,500
Spring Creek	Hatchery	no	-	yes	8,200
L. Bonneville	Hatchery	no	-	yes	26,200
Cowlitz Fall	Hatchery	no	-	yes	8,800
Willamette Spring	Hatchery	no	-	yes	13,500
Cowlitz Spring	Hatchery	no	-	yes	2,500

3.2. Lewis River Fall Chinook Salmon

3.2.1. Summary

A stock-recruit analysis for the Lewis River fall chinook salmon stock was conducted. Estimates of spawning escapements and resulting recruitments for brood years 1964 to 1991 were fit with a simple Ricker model to derive scientific estimates of the maximum sustained yield escapement level and maximum sustained yield exploitation rate. The adjusted R^2 in the stock recruit relationship developed was 0.56. Estimates of the maximum sustained yield escapement level and the maximum sustained yield exploitation rate with 90% confidence intervals along with estimated parameters in the model are summarized next.

Year Classes Used in Analysis	$\hat{S}_{MSY}$ (90% C.I.)	$\ln \hat{\alpha}$	$\hat{\beta}$	$\hat{U}_{MSY}$ (90% C.I.)
1964 – 1991	5,791[a] (4,950 – 7,076)	2.1892	0.000131	0.76[b] (0.66 – 0.83)

[a] Because this estimate has not been adjusted for process error or measurement error in estimated production or spawning abundance, it is probably biased low ($\hat{S}_{MSY} < S_{MSY}$).

[b] Because this estimate has not been adjusted for process error or measurement error in estimated production or spawning abundance, it is probably biased high ($\hat{U}_{MSY} > U_{MSY}$).

At this time, the CTC concludes that 5,791 age-3 to 6 fish in the escapement is the best estimate for S_{MSY} for the Lewis River fall chinook salmon stock, even though this estimate should be viewed as potentially biased. Estimates of measurement error in statistics on production and spawning escapement were not available, and therefore could not be included in the analysis. As a result, estimates of S_{MSY} and U_{MSY} were calculated with a "naive" analysis (no corrections for measurement and process error). Likely, the estimate of S_{MSY} is low and the estimate of U_{MSY} is high ($\hat{S}_{MSY} < S_{MSY}$ and $\hat{U}_{MSY} > U_{MSY}$). The potentially biased estimate for U_{MSY} is 0.76. The potentially biased estimate for S_{MSY} of 5,791 is similar to an estimate of 5,700 established by McIsaac (1990) on data for year classes through 1982.

<u>Primary References:</u>

Hawkins, S. 1996. Results of sampling the Lewis River natural spawning fall chinook population in 1995. WDFW Col. R. Progress Report. 96-06. April, 1996. 10 pp.

McIsaac, Donald O. 1977. Total spawner population estimate for the North Fork Lewis River based on carcass tagging, 1976. Washington Department of Fisheries, Columbia River Laboratory Progress Report Number 77-1. Battleground, Washington.

Schaller, H., O. Langness, P. Budy, E. Tinez. 1998. FY98 Final Report. Section 3.1.2 Run Reconstruction Information. PATH.

3.2.2. Stock Description and Stock Assessment Data

Stock Description: Lewis River fall chinook salmon (LRF) comprise the majority of natural fall production below Bonneville Dam, and over 80% (on average) of the Lower River Wild Management Unit (small amounts of natural production from the Cowlitz and Sandy River basins are included in this unit). The Lewis River fall stock smolt as sub-yearlings and hence, have an ocean-type life history pattern. These fish return at ages 2-6, and spawn some time after returning to natural areas (brights). Juveniles rear in areas with flows regulated by upstream dams. Adults generally begin freshwater migration in early August with peak spawning in mid November.

In 1931, construction of Merwin Dam (Lewis River Kilometer (Rkm) 31.4) blocked migrating adults from at least one-half of their historic spawning habitat. The main spawning area is now the 6.4 km below Merwin Dam and above Lewis River Hatchery (Rkm 25.3). Most juveniles rear below the hatchery and above a low falls (Rkm 11). Habitat quality is considered to be fair to good.

Hatchery Influence: Most chinook salmon returning to the Lewis River are wild fish with a few hatchery strays. Abundance of hatchery strays is estimated by expanding the number of coded-wire tags (CWTs) recovered in the North Fork of the Lewis River by sampling and marking rates. Four hatcheries have consistently represented the majority of strays to the North Fork during the 1979 to 1997 return years: Lewis Hatchery Complex (Lewis Hatchery and Speelyai Hatchery); Washougal Hatchery; Cowlitz Salmon Hatchery; and the Kalama Hatchery Complex (Fallert Creek [a.k.a., Lower Kalama] Hatchery, and Kalama Falls Hatchery). Lewis River and Speelyai hatcheries on the North Fork have intermittently released fingerlings from returns to the Merwin Dam fish trap. These releases have not exceeded 550,000 fingerlings and typically ranged from 50,000 to 150,000 fish.

Escapement: Annual estimates of the number of spawners in the Lewis River were obtained by expanding peak counts from weekly counts of live and dead fish (brights and tule) in the 6.4-km area below Merwin Dam (Rkm 31.4) (McIsaac 1990, citing Norman 1988) by the ratio of 5.2685 (total spawners/peak count). The expansion ratio resulted from a 1976 carcass tagging and recapture study (McIsaac 1977). Most in-river spawning occurs above the Lewis Hatchery (Rkm 25.3), though spawners are found downstream to Rkm 18.5 and in the lower reaches of some tributaries (e.g. Cedar Creek). Methods of recovery, counting, and expansion of the index area fish have been consistent since 1964. All naturally spawning fish, both hatchery and natural production, are included in the estimated abundance of the spawning population.

Table 3.2 provides estimates of spawning escapement and estimated production for the Lewis River fall chinook salmon stock. Contrast in estimated escapements ranged from 3,371 in 1976 to 21,199 in 1989, a 6.3 fold level of variation. At this time, no information is compiled on the sex ratio of adults or the sample sizes associated with annual estimated age compositions.

Production: Coded-wire tags recovered from the 1977-1979 year classes of wild fish were used to estimate catches directly for those year classes and indirectly for others. The average ratio of (catches + spawners) to spawners for the 1977-1979 classes was used to expand spawners to total

recruitment (production) for prior year classes. Hatchery strays were subtracted from the estimate of spawning abundance prior to the expansion. Hawkins (1998) provides age specific estimates of spawning abundance for 1964-1997 which were used to estimate production by age group and subsequently by year class. Age 3-6 fish were considered as spawners because age-2 fish are almost exclusively male.

Expansions to estimate production have been stratified by source of exploitation. If the estimated number of annual recruits to the LRF(s) at the mouth of the Columbia River is identified as:

$$ColRecruits_{i,y} = \frac{UpRecruits_{i,y}}{\left(\left(1 - MainExp_{i,y+i}\right) * \left(1 - TribExp_{i,y+i}\right)\right)},$$

where i is age, y is year class, $MainExp_{i,j}$ is an estimate of the exploitation rate in the main-stem of the Columbia River for fish age i in year j (note $j = y + i$), *TribExp* is the estimate of the exploitation rate in the tributary, and *UpRecruits* is the estimated number of wild fish spawning in the Lewis River. Exploitation rates in tributaries pertain to the sport fishery in the North Fork of the Lewis River; exploitation rates in the main-stem Columbia River were estimated for commercial and sport fisheries from return of CWTs representing the LRF(s). Prior to 1980, main-stem exploitation rates were estimated from catch and run size data for Lower Columbia River fall chinook salmon. For 1964-68, average estimated sport exploitation rates were assumed. Upriver recruits were estimated as follows:

$$UpRecruits_{i,y} = \left(Spawners_{y+i} + TrapFish_{y+i} - HatSpawn_{y+i}\right) * AgeProp_{i,y+i},$$

where *Spawners* is the estimated number of spawners (expanded from peak counts), *TrapFish* is the number of natural origin fish removed prior to spawning for artificial production programs, i.e., not accounted for in peak count estimate, *HatSpawn* is the estimated number of hatchery fish spawning naturally (strays), and *AgeProp* is the estimated proportion of fish at age for a given return year.

Table 3.2. Estimated spawning escapements and estimated production used to determine a biologically based escapement goal associated with *MSY* for the Lewis River fall stock of chinook salmon.

Brood Year	**Estimated Escapement**	**Estimated Production**
1964	16,857	38,738
1965	7,927	11,324
1966	11,627	21,045
1967	9,711	46,887
1968	7,160	72,837
1969	4,986	25,305
1970	4,130	19,572
1971	19,926	35,638
1972	18,488	16,616
1973	9,120	17,280
1974	7,549	14,776
1975	13,859	20,082
1976	3,371	30,360
1977	6,930	30,948
1978	5,363	10,551
1979	8,023	22,687
1980	16,394	12,941
1981	19,297	13,591
1982	8,370	22,497
1983	13,540	33,777
1984	7,132	43,902
1985	7,491	32,086
1986	11,983	24,225
1987	12,935	12,476
1988	12,059	14,842
1989	21,199	2,489
1990	17,506	25,417
1991	9,066	11,219

Production (*TotRecruits*) by age in a year class includes ocean harvest impacts (in adult equivalents), and is estimated as follows (Deriso 1998):

$$TotRecruits_{i,j} = \frac{ColRecruits_{i,j}}{\prod_{k=2}^{i}(1 - OCNExp_k)}$$

where *OCNExp* is the estimated ocean exploitation rate (see following), and *k* is the first age fish are vulnerable to ocean fishing. Table 3.2 contains estimates of production for year classes 1964 through 1991.

Estimation of Ocean Exploitation Rates: Ocean exploitation rates were estimated from CWT data using the backward cohort method currently used by the CTC (CTC 1987, see Table 3). The cohort size at any age includes all mortalities which occur in that year plus the number of fish alive at the end of that fishing year (cohort size at age is increased for natural mortality after fishing mortalities have been included). The cohort size is first estimated from the total of all the legal catches and escapement. Incidental mortalities (shaker mortalities and mortalities in non-retention fisheries) are then estimated iteratively from the legal catch cohort size and added back into the cohort. Natural mortality in the ocean is assumed to be constant in this analysis and is set at 0.4 for age 2, 0.3 for age 3, 0.2 for age 4, and 0.1 for ages 5 and 6. Ocean exploitation rates were calculated as the total ocean fishing mortality (ocean catch + incidental fishery impacts) divided by the cohort size less natural mortality. River mouth recruits were expanded by the cumulative exploitation rate by age for a year class. No CWT data are available for year classes before 1977. To estimate ocean exploitation rates for these early years, a historic fishery index was first calculated as the ratio of the catch per unit effort in each year to the average catch per unit effort in base years 1979 to 1982. Secondly, catch of LRF(s) in each fishery was estimated by multiplying the average catch distribution for LRF(s) during the base period (catch years 1979-1982) by the historical fishery index for each year. The basic cohort and exploitation analysis described above was then completed for the years prior to availability of CWT data.

3.2.3. Stock-Production Analysis.

Measurement Error: No sampling variances are available for estimates of spawning abundance, nor have any variances been calculated for estimates of recruitment (production).

Parameter Estimates: Paired estimates of spawning abundance and resulting production for chinook salmon spawning in the Lewis River (year classes 1964-1991) were fit to Ricker's model (Ricker 1975) with an assumed multiplicative error term:

$$R_y = \alpha S_y \exp(-\beta S_y + \varepsilon_y),$$

where S_y is the number of spawning chinook salmon that produced year class y, R_y is the recruitment (production) associated with year class y, α and β are the density-independent and density-dependent parameters, and ε_t is process error distributed normally with mean 0 and variance σ^2_ε. Ricker's model was linearized and estimates of S_y and R_y substituted into the relationship to create:

$$\ln(\hat{R}_y / \hat{S}_y) = \ln(\alpha) - \beta\hat{S}_y + r_y,$$

where r_y is distributed normally with mean 0 and variance σ^2_r. Variance σ^2_r is the sum of process error and measurement error in estimates of production ($\sigma^2_r = \sigma^2_\varepsilon + \sigma^2_v$ where σ^2_v is a function of variances resulting from sampling to estimate production). While measurement error in estimates of spawning abundance and production is a part of the data, there is no expression for the corresponding variance in the formulation above. The model was fit to paired estimates with the statistical package SAS™ PROC REG. Results of these analyses are summarized in Table 3.3. Estimates of production and spawning abundance by year class along with the fitted curve are

Estimation of Ocean Exploitation Rates: Ocean exploitation rates were estimated from CWT data using the backward cohort method currently used by the CTC (CTC 1987, see Table 3). The cohort size at any age includes all mortalities which occur in that year plus the number of fish alive at the end of that fishing year (cohort size at age is increased for natural mortality after fishing mortalities have been included). The cohort size is first estimated from the total of all the legal catches and escapement. Incidental mortalities (shaker mortalities and mortalities in non-retention fisheries) are then estimated iteratively from the legal catch cohort size and added back into the cohort. Natural mortality in the ocean is assumed to be constant in this analysis and is set at 0.4 for age 2, 0.3 for age 3, 0.2 for age 4, and 0.1 for ages 5 and 6. Ocean exploitation rates were calculated as the total ocean fishing mortality (ocean catch + incidental fishery impacts) divided by the cohort size less natural mortality. River mouth recruits were expanded by the cumulative exploitation rate by age for a year class. No CWT data are available for year classes before 1977. To estimate ocean exploitation rates for these early years, a historic fishery index was first calculated as the ratio of the catch per unit effort in each year to the average catch per unit effort in base years 1979 to 1982. Secondly, catch of LRF(s) in each fishery was estimated by multiplying the average catch distribution for LRF(s) during the base period (catch years 1979-1982) by the historical fishery index for each year. The basic cohort and exploitation analysis described above was then completed for the years prior to availability of CWT data.

3.2.3. Stock-Production Analysis.

Measurement Error: No sampling variances are available for estimates of spawning abundance, nor have any variances been calculated for estimates of recruitment (production).

Parameter Estimates: Paired estimates of spawning abundance and resulting production for chinook salmon spawning in the Lewis River (year classes 1964-1991) were fit to Ricker's model (Ricker 1975) with an assumed multiplicative error term:

$$R_y = \alpha S_y \exp(-\beta S_y + \varepsilon_y),$$

where S_y is the number of spawning chinook salmon that produced year class y, R_y is the recruitment (production) associated with year class y, α and β are the density-independent and density-dependent parameters, and ε_t is process error distributed normally with mean 0 and variance σ_ε^2. Ricker's model was linearized and estimates of S_y and R_y substituted into the relationship to create:

$$\ln(\hat{R}_y / \hat{S}_y) = \ln(\alpha) - \beta \hat{S}_y + r_y,$$

where r_y is distributed normally with mean 0 and variance σ_r^2. Variance σ_r^2 is the sum of process error and measurement error in estimates of production ($\sigma_r^2 = \sigma_\varepsilon^2 + \sigma_v^2$ where σ_v^2 is a function of variances resulting from sampling to estimate production). While measurement error in estimates of spawning abundance and production is a part of the data, there is no expression for the corresponding variance in the formulation above. The model was fit to paired estimates with the statistical package SAS™ PROC REG. Results of these analyses are summarized in Table 3.3. Estimates of production and spawning abundance by year class along with the fitted curve are presented in Figure 3.1. Plots of residuals, autocorrelation functions, and partial autocorrelation functions indicate no statistically significant autocorrelation (Figure 3.2). The spawning abundance that on average produces maximum sustained yield (S_{MSY}) was estimated by solving the following relationship from Ricker (1975: p. 347, Model 1, entry 17):

$$1 = (1 - \beta S_{MSY})\exp[\ln(\alpha) - \beta S_{MSY}]$$

with substitutions $\ln(\hat{\alpha}) \rightarrow \ln(\alpha)$ and $\hat{\beta} \rightarrow \beta$. The adjustment in $\hat{S}_{MSY}$ for process error (see Section 1.6.1) was omitted because the correction could not be isolated without a corresponding estimate for measurement error in estimated production. Nor was the analysis adjusted for measurement error in estimates of spawning abundance. The estimate of S_{MSY} of 5,791 derived from this analysis is essentially the same as the escapement goal of 5,700 used previously by the CTC and initially developed by McIsaac (1990) based on spawner-recruit analysis of data for the 1964-1982 year classes. An estimate $\hat{U}_{MSY}$ for the exploitation rate associated with S_{MSY} was calculated as $\hat{U}_{MSY} = \hat{\beta}\hat{S}_{MSY}$ = 0.76.

Table 3.3. Estimated parameters for regression on the log-linear transform of Ricker's model on estimates of production and spawning abundance of chinook salmon in the Lewis River.

$\ln(\hat{\alpha})$	2.1892 (P < 0.0001)
$\hat{\beta}$	0.000131 (P< 0.0001)
$\hat{\sigma}_r^2$	0. 3739
$\hat{\sigma}_\varepsilon^2$	?
R^2(corrected)	0.56
$\hat{S}_{MSY}$	5,791
$\hat{U}_{MSY}$	0.76

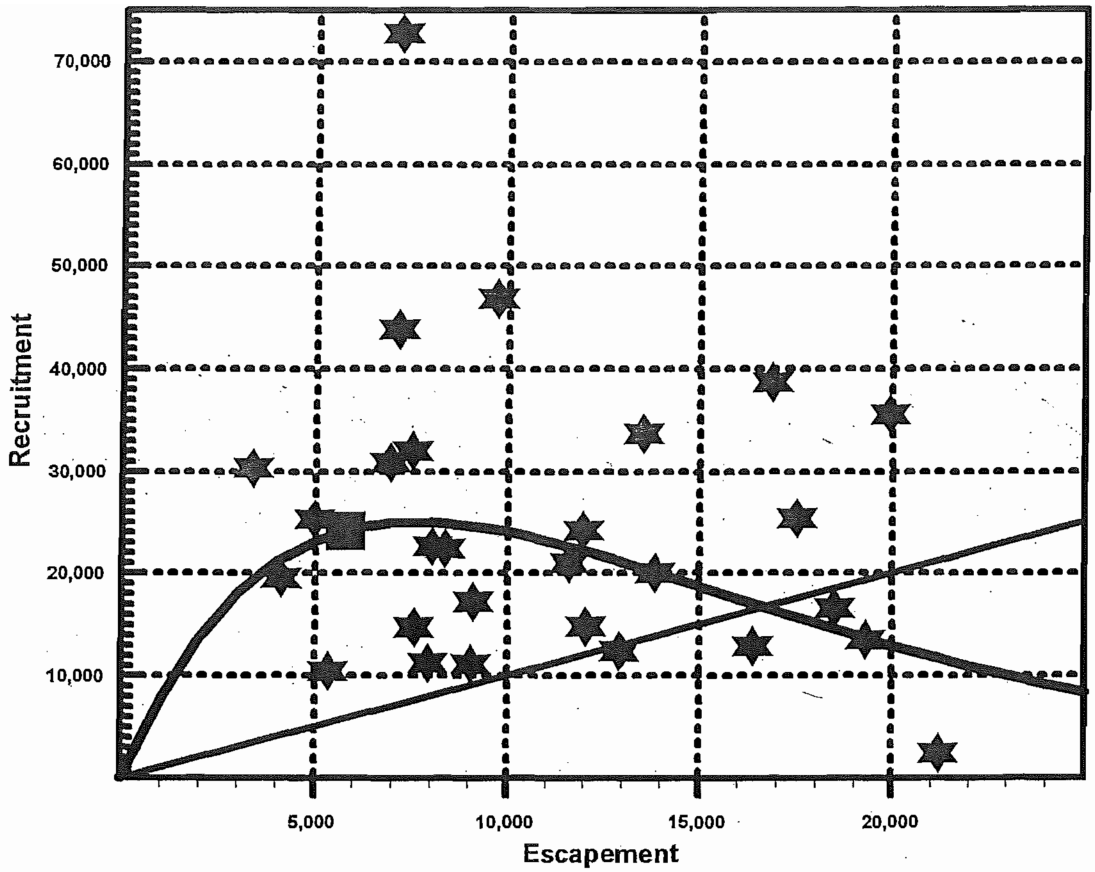
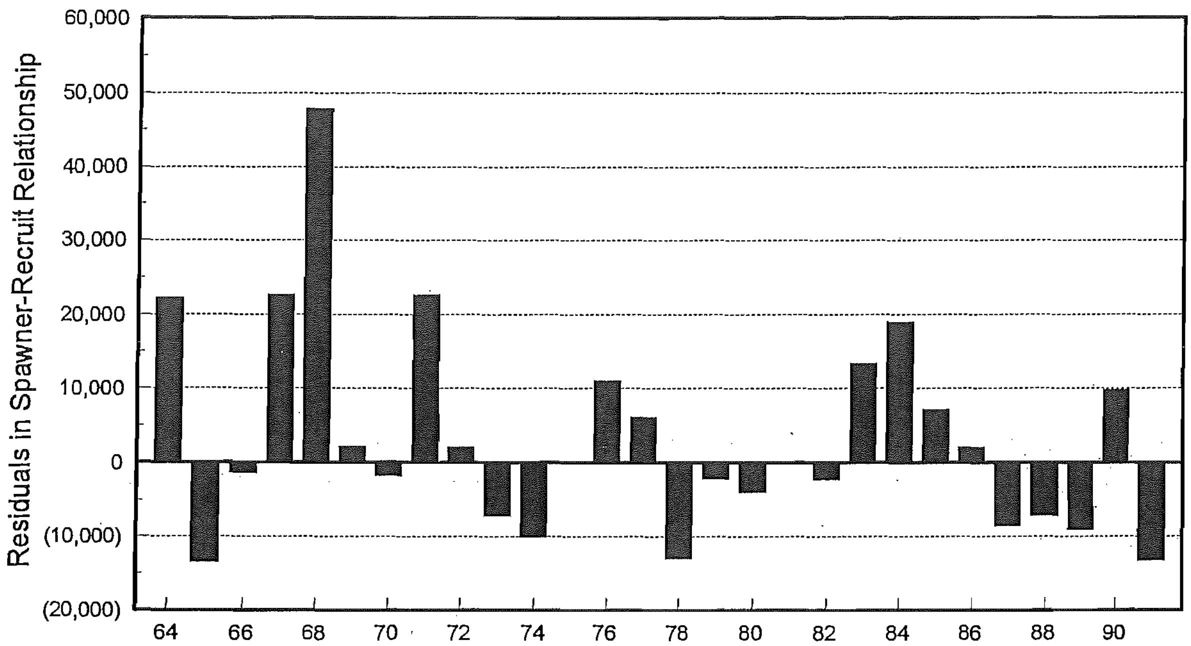


Figure 3.1. Predicted recruitment as a function of spawning escapement for the Lewis River fall chinook salmon stock (curved line) with stars representing specific brood year spawning escapements and resultant recruitments estimated for year classes 1964 through 1991. The solid square corresponds to predicted recruitment and spawning escapement when escapement is 5,791 spawners.



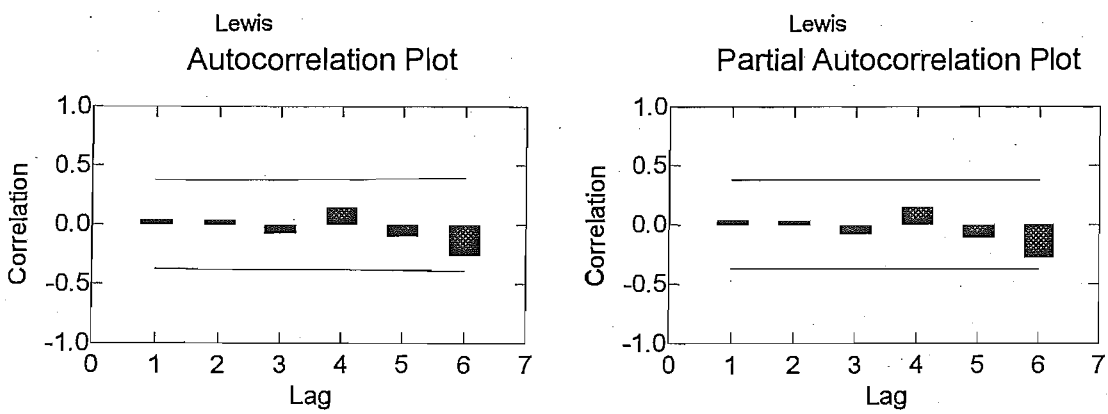


Figure 3.2. Time plot of residuals in the spawner-recruit relationship for the Lewis River fall chinook salmon stock (upper panel) and lag plots of autocorrelation and partial autocorrelation functions for the same relationship (lower panel).

Table 3.4. Statistics from bootstrap simulations to estimate standard errors and 90% confidence intervals for S_{MSY} and U_{MSY} for the Lewis River fall chinook salmon stock.

	Estimate	SE	Bootstrap Mean	Bias	90% lower limit	90% upper limit
S_{MSY}	5,791	706	5,864	74	4,950	7,076
U_{MSY}	0.76	0.05	0.75	0.01	0.66	0.83

Simulation. The estimated 90% confidence intervals for $\hat{S}_{MSY}$ and $\hat{U}_{MSY}$ (Table 3.4) were calculated through bootstrap simulation of residuals in the fit of Ricker's model (see Section 1.8.2 for description of methods). One thousand new data sets were generated from the original by adding residuals drawn with replacement to values predicted with Ricker's model, then Ricker's model was refit to each set to produce new parameter estimates. No adjustment was made for measurement error in spawning abundance. Estimated statistical bias was negligible in estimates of both S_{MSY} and U_{MSY}.

3.2.4. Discussion

Although the CTC concluded that 5,791 spawners is the best biologically based escapement goal for the Lewis River fall stock of chinook salmon estimable at this time, the estimate is likely biased low. The bias arises from being unable to adjust the analysis for process and measurement error. Measurement error arises from substituting estimates of recruitment and especially spawning abundance for actual values when fitting stock-production models. Measurement error in production obscures the effect of process error. Measurement error in spawning abundance biases the estimates of S_{MSY} downward and U_{MSY} upward, making the stock appear more productive than it is (Hilborn and Walters 1992:288).

Although no estimates of measurement error were presented for the data used in this analysis, such measurement error obviously exists. For instance, all escapements were based on expansions of peak counts by a factor of 5.2685 from the 1976 study. Multiple estimates of the expansion factor would be needed to estimate the measurement error associated with this data manipulation. Another common source of sampling error occurs with estimating age composition and harvest based on expanding information from the return of CWTs. Although no sampling variances were estimated or used in this analyses, the sampling error they represent is present, but not quantified.

Comparing the bootstrap mean with the point estimate shows little statistical bias in estimates of S_{MSY} and U_{MSY}, however, this estimate of statistical bias is predicated on the number of spawners and production being known, not estimated as is the situation here.

3.3. Columbia Upriver Summer Chinook Salmon

3.3.1 Summary

A stock-recruit analysis for the Columbia upriver summer chinook salmon stock was conducted. Values for spawning escapements and resulting recruitments for brood years 1979 to 1995 were based upon predictions obtained from CTC chinook model calibration 98-12. Analyses used these predicted values rather than actual estimates because data concerning brood year specific spawning escapements and resulting recruitments were not available. These predicted values were fit with a simple Ricker model. Lack of contrast in predictions of spawning abundance (max/min predictions = 2.5) cast doubt on the accuracy of estimates derived from the analysis. Lack of knowledge on the accuracy and precision of predictions from the CTC chinook model probably results in the estimated maximum sustained yield escapement estimate being biased low and the estimated maximum sustained yield exploitation rate being biased upward ($\hat{S}_{MSY} < S_{MSY}$ and $\hat{U}_{MSY} > U_{MSY}$). Autocorrelation observed in the analysis is exclusively a result of using predictions from the CTC chinook model; whether such autocorrelation occurs across production or spawning abundance estimated from field studies is unknown. The adjusted R^2 in the stock recruit relationship developed was 0.66. Estimates of the maximum sustained yield escapement level and the maximum sustained yield exploitation rate with 90% confidence intervals along with estimated parameters in the model are summarized below.

Year Classes 1979 – 1995[a]	$\hat{S}_{MSY}$ (90% C.I.)	$\ln \hat{\alpha}$	$\hat{\beta}$	$\hat{U}_{MSY}$ (90% C.I.)
Past Rock Island Dam	12,143[b] (8,847 – 15,981)	2.1516	0.000062	0.75[c] (0.71 – 0.82)
Past Bonneville Dam	17,857[b] (13,010 – 23,501)	"	"	0.64[c] (0.60 – 0.70)

[a] Data for this analysis were derived from predictions taken from CTC chinook model calibration 98-12.

[b] Because this estimate has not been adjusted for process error or measurement error in estimated production or spawning abundance, it is probably biased low ($\hat{S}_{MSY} < S_{MSY}$).

[c] Because this estimate has not been adjusted for process error or measurement error in estimated production or spawning abundance, it is probably biased high ($\hat{U}_{MSY} > U_{MSY}$).

At this time, the CTC proffers 12,143 fish past Rock Island Dam and 17,857 fish past Bonneville Dam as interim escapement goals for the Columbia upriver summer stock of chinook salmon even though these estimates should be viewed as likely biased. Analyses establishing these goals were not based on field data and resulting estimates, but on predictions from the CTC chinook model. For this reason, these interim goals should not be considered as biologically based. However, the CTC will use these interim goals until a more thorough analysis based upon detailed stock-production information is completed by the Washington Department of Fish and Wildlife and reviewed by the CTC. The final report for this study is scheduled for fall of 2000 and the CTC should be able to reach agreement upon a biologically based goal for the Columbia upriver summer stock of chinook salmon during the spring of 2001.

3.3.2 Stock Description and Stock Assessment Data

In CTC assessments of spawning escapements, 85,000 adult chinook salmon counted at Bonneville Dam from June 1 to July 31 has been used as a rebuilding goal. This number and goal includes hatchery production and natural production. Naturally produced chinook salmon passing Bonneville Dam at this time of year are largely composed of two groups, those fish destined for the Snake River watershed and those fish destined for the upper waters of the Columbia River watershed. Summer chinook salmon that spawn in the Snake River migrate to sea primarily as yearlings (stream type) and are rarely landed in ocean fisheries. Summer chinook that spawn in the upper portions of the Columbia River, however, migrate to sea primarily as fingerlings (ocean type) and are caught in ocean fisheries in substantial numbers.

Summer chinook salmon destined for the Snake River migrate upstream through four mainstem Columbia River dams: (1) Bonneville completed in 1938, (2) Dalles completed in 1957, (3) John Day completed in 1968, and (4) McNary completed in 1953. After entering the Snake River, these fish have to pass four mainstem Snake River dams: (1) Ice Harbor completed in 1961, (2) Lower Monument completed in 1969, (3) Little Goose completed in 1970, and (4) Lower Granite completed in 1975. These fish eventually migrate into tributaries of the Snake River above Lower Granite Dam.

Summer chinook that spawn in the Columbia River upstream of its confluence with the Snake River represent a stock that once migrated well past Grand Coulee Dam (completed in 1941 and constructed without fish passage facilities). Historically, these fish spawned mostly in Canada in the headwaters of the Columbia. As part of the federal Grand Coulee fish maintenance project, fish were trapped at the base of Grand Coulee Dam in the years 1939 to 1941 and were transplanted into hatcheries or into the Okanogan, Methow, and Wenatchee rivers to spawn naturally. Downstream fences were installed in these rivers to prevent chinook from migrating back out of these rivers. Currently, all Columbia upriver summer chinook migrate through seven mainstem dams: (1) Bonneville - completed in 1938, (2) Dalles - completed in 1957, (3) John Day - completed in 1968, (4) McNary - completed in 1953, (5) Priest Rapids Dam - completed in 1959, (6) Wanapum Dam - completed in 1963, and (7) Rock Island Dam - completed in 1933. Some Columbia upriver summer chinook migrate into the Wenatchee River to spawn with the rest migrating further upstream passing Rocky Reach Dam (completed in 1961) and Wells Dam (completed in 1967) and eventually spawning in the Methow and Okanogan Rivers.

Although chinook salmon spawning in the Snake River and the upper Columbia River are distinct stocks, they migrate together as returning adults. Management agencies in the Columbia River have agreed upon an inriver fisheries management goal of 80,000 to 90,000 chinook salmon passing Bonneville Dam between June 1 and July 31. The management range of 80,000 to 90,000 was based upon analysis of data from return years 1938 to 1958. The management goal reportedly considered an increasing sport fishery in headwater areas and increasing losses of migrating adults at various dams. The CTC has used the mid-point of this range (85,000) as an escapement goal for the Columbia upriver summer stock of chinook salmon.

Besides the mixed-stock nature of the management goal, habitat has been degraded since the data upon which it was established were collected. In 1938 only one of the dams listed above was in place, from then through 1958 an additional three main-stem dams were completed, and since 1958, nine more main-stem dams have been constructed. Even if the original analysis resulted in appropriate escapement goals for these stocks during the late 1950s, there is good reason to believe appropriate S_{MSY} escapement goals today would be quite different due to development that has taken place since then.

A separate escapement goal should be developed for the Columbia upriver summers. Because of differences in productivity related to differences in life history, the Columbia upriver summer indicator stock is not biologically relevant to the Snake River stock. For this reason, the CTC in 1997 partitioned the escapement goal at Bonneville Dam into a goal for Columbia upriver summers and a goal for the Snake River summer run. Counts of chinook salmon at Priest Rapids Dam (first dam located on the Columbia River upstream of its confluence with the Snake River) and at Ice Harbor Dam (first dam located in the Snake River) from 1964 to 1969 show that on average 44.5% of the fish migrated into the Upper Columbia River while 55.5% of the fish migrated into the Snake River. The estimate of 44.5% was multiplied by the lower bound of the overall escapement goal of 80,000 and resulted in a model escapement goal of 35,600 for Columbia upriver summer chinook. Although this provided a more appropriate goal for modeling, it was not an effort to determine a biologically based escapement goal for Columbia upriver summer chinook salmon.

In 1996 the CTC was assigned the task of estimating an appropriate MSY escapement goal for Columbia upriver summer chinook; the task has not been completed. To complete the task, a detailed set of data including annual escapements and recruits resulting from these escapements is needed. This task is fairly complicated for the summer stocks of chinook salmon in the Columbia River because of: (1) inter-dam loss of adults and stock specific estimates of these rates, (2) the lack of precise spawning ground escapement estimates, (3) the general lack of age composition estimates of spawners and those fish caught in fisheries, and (4) the general lack of fishery exploitation rates. However, some progress has been made and it is anticipated that a thorough analysis leading to the identification of an appropriate MSY escapement goal can be achieved with some additional effort.

The remainder of this chapter provides a description of how an interim goal was determined for the Columbia upriver summer stock of chinook salmon. The CTC decided to use predictions from CTC chinook model calibration 98-12 as surrogates for estimates of production and spawning abundance. While the result would not be a biologically based escapement goal, it would hopefully be comparable against a goal of 35,600 spawners used in modeling and a more useful statistic than the out-dated rebuilding goal of 85,000 past Bonneville Dam (Table 3.1).

<u>CTC Chinook Model Data:</u> CTC chinook model calibration 98-12 was used to generate estimates of escapement and adult equivalent cohort size for brood years 1979 to 1995. This time frame occurred four years after the last main-stem dam was completed and well after major development of most of the drainage occurred. A summary of inter-dam conversions from Bonneville Dam to Rock Island Dam (the last dam the naturally spawning fish returning to the

Wenatchee River have to pass) was computed and applied to the escapements passed Bonneville Dam as predicted by the model. These model predictions are summarized in Table 3.5.

3.3.3. Stock-Production Analysis

Measurement Error: No estimated variances are available for model predictions of spawning abundance, nor have any variances been calculated for predictions of recruitment (production).

Parameter Estimates: Paired predictions of spawning escapements and resulting recruitments for Columbia upriver summer chinook from CTC model calibration 98-12 were fit to a Ricker curve (Ricker 1975) with an assumed multiplicative error term of the following form:

$$R_y = \alpha S_y \exp(-\beta S_y + \varepsilon_y),$$

where y is the year (class), S_y is the number of spawning chinook salmon that produced year class y, R_y is the recruitment associated with year class y, α and β are the density-independent and density-dependent parameters, and ε_y is distributed normally with mean 0 and variance σ_ε^2. The Ricker model used was linearized as follows:

$$\ln(\tilde{R}_y / \tilde{S}_y) = \ln(\alpha) - \beta \tilde{S}_y + \varepsilon_y,$$

where variables with "~" are predictions. The model was fit to paired predictions with the statistical package SAS™ PROC REG. Results of these analyses are summarized below:

n	$\ln(\hat{\alpha})$	$SE(\ln(\hat{\alpha}))$	$\hat{\beta}$	$SE(\hat{\beta})$	Adjusted R^2
17	2.31163	0.27446	0.00007447	0.0000205	0.433

Plots of residuals, autocorrelation functions, and partial autocorrelation functions show statistically significant autocorrelation at a lag of one generation (Figure 3.3). The cyclical decay in the autocorrelation function and the random values of the partial autocorrelation function after the first lag suggests that an auto-regressive AR(1) model is the appropriate model (see Section 1.7.2). Ricker's stock-recruit model with auto-regressive error with lag one generation is:

$$\ln(R_y / S_y) = ln(\alpha) + \beta S_y + a_y / (1 - \phi_1 \mathrm{B}),$$

where B is the back-shift operator and a_y is "white noise" with mean 0 and variance σ_a^2. Multiplying both sides of the equation above by $(1 - \phi_1 \mathrm{B})$, inserting predictions from the CTC model, and simplifying gives:

$$\ln(\tilde{R}_y / \tilde{S}_y) = (1 - \phi_1)\ln(\alpha) + \phi_1 \ln(\tilde{R}_{y-1} / \tilde{S}_{y-1}) - \beta(\tilde{S}_y - \phi_1 \tilde{S}_{y-1}) + r_y,$$

where r_y are the residuals (see Section 1.7.2). Fitting this model to these predictions with SAS™ PROC ARIMA produced the following estimates for parameters:

N	$\hat{\ln(\alpha)}$	SE($\hat{\ln(\alpha)}$)	$\hat{\beta}$	SE($\hat{\beta}$)	Adjusted R^2
17	2.1516	0.24246	0.0000620	0.0000129	0.66

Plots of autocorrelation and partial autocorrelation functions indicated no autocorrelation among residuals from the fitted AR(1) model. The spawner-recruit relationship is presented in Figure 3.4.

The spawning abundance that on average produces maximum sustained yield (S_{MSY}) was estimated by solving the following relationship from Ricker (1975: p. 347, Model 1, entry 17):

$$1 = (1 - \beta S_{MSY})\exp(\ln\alpha)\exp(-\beta S_{MSY}),$$

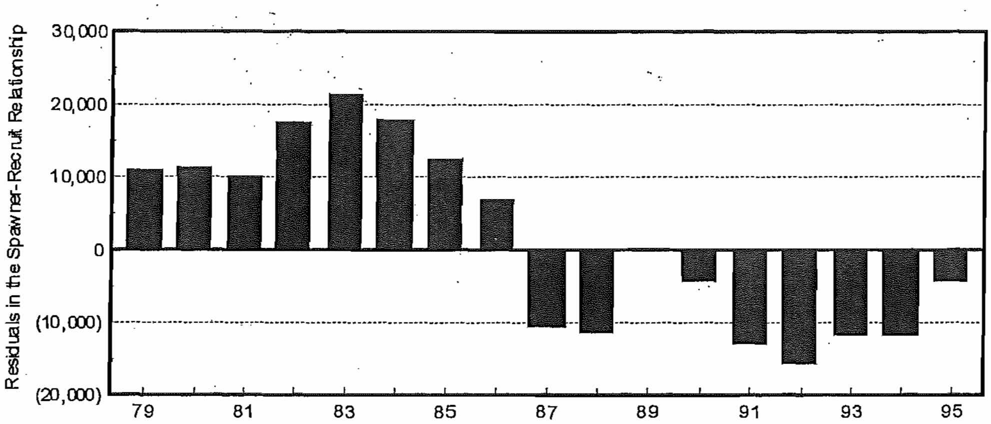
with substitutions $\hat{\ln(\alpha)} \rightarrow \ln(\alpha)$ and $\hat{\beta} \rightarrow \beta$. The same equation can be used for both the traditional version of Ricker's model and the AR(1) version because ϕ_1 is solely part of the deviation in both formulations ($\varepsilon_y = a_y - \phi_1 a_{y-1}$ in the AR(1) version). In both instances, the associated exploitation U_{MSY} was estimated as $\hat{U}_{MSY} = \hat{\beta}\hat{S}_{MSY}$. The subsequent results are listed in the table below for both versions of Ricker's model:

	ESTIMATES	
Parameter	**Traditional Ricker Model**	**AR(1) Ricker Model**
$\ln(\alpha)$	2.3124	2.1516
β	0.00007447	0.0000620
ϕ	-	0.808
MSY	37,958	37,041
S_{MSY} (SE) 90% CI	10,517 (3,797) 7,978-17,567	12,143 (1,961) 8,847-15,981
U_{MSY} (SE) 90% CI	78% (5.4%) 68%-85%	75% (3.5%) 71%-82%

Simulation. A non-parametric bootstrap re-sampling algorithm was used to estimate 90% confidence intervals for $\hat{S}_{MSY}$ and $\hat{U}_{MSY}$ (see Section 1.8.2). Residuals from both versions of Ricker's model were re-sampled, then added to predictions to produce new sets of "observed" values for dependent variables. Each new set of values was regressed against original values for spawning abundance to produce a new set of parameter estimates and subsequently new estimates for S_{MSY} and U_{MSY}. One thousand new data sets, 1,000 sets of parameter estimates, and 1,000 paired estimates $\tilde{S}_{MSY}$, $\tilde{U}_{MSY}$ were so generated. Standard error for $\hat{S}_{MSY}$ was estimated as:

$$se(\hat{S}_{MSY}) = \sqrt{\frac{\sum_{k=1}^{1000}(\tilde{S}_{MSY(k)} - \bar{S}_{MSY})^2}{1000-1}},$$

where $\bar{S}_{MSY} = (1000)^{-1}\sum_{k=1}^{1000}\tilde{S}_{MSY(k)}$. The estimate $se(\hat{U}_{MSY})$ was similarly calculated. The percentile method (see Section 1.8.2) was used to provide 90% confidence intervals about $\hat{S}_{MSY}$ and about $\hat{U}_{MSY}$. Confidence intervals and standard errors for estimates for both versions of Ricker's model are reported in the previous table. A comparison of the bootstrap mean with the point estimate showed little statistical bias, but additional bias might be present due to measurement error in the data.



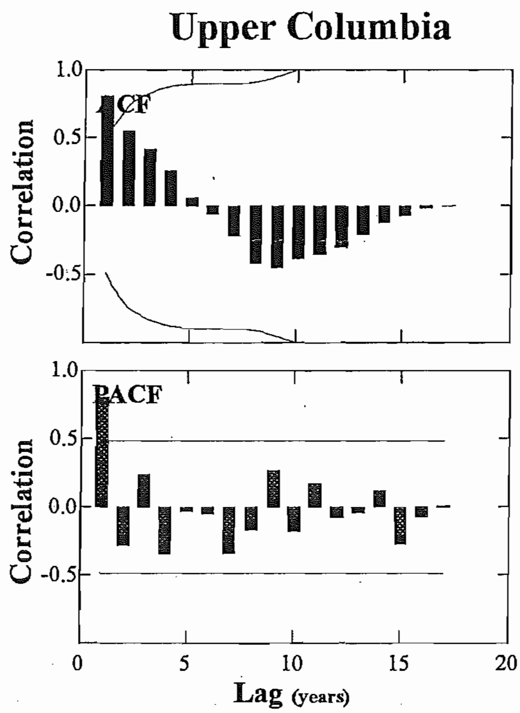


Figure 3.3. Plot of residuals in the spawner-recruit relationship for the upper Columbia River summer chinook salmon stock (upper panel) and plot of autocorrelation and partial autocorrelation in the stock-recruit relationship (lower panel).

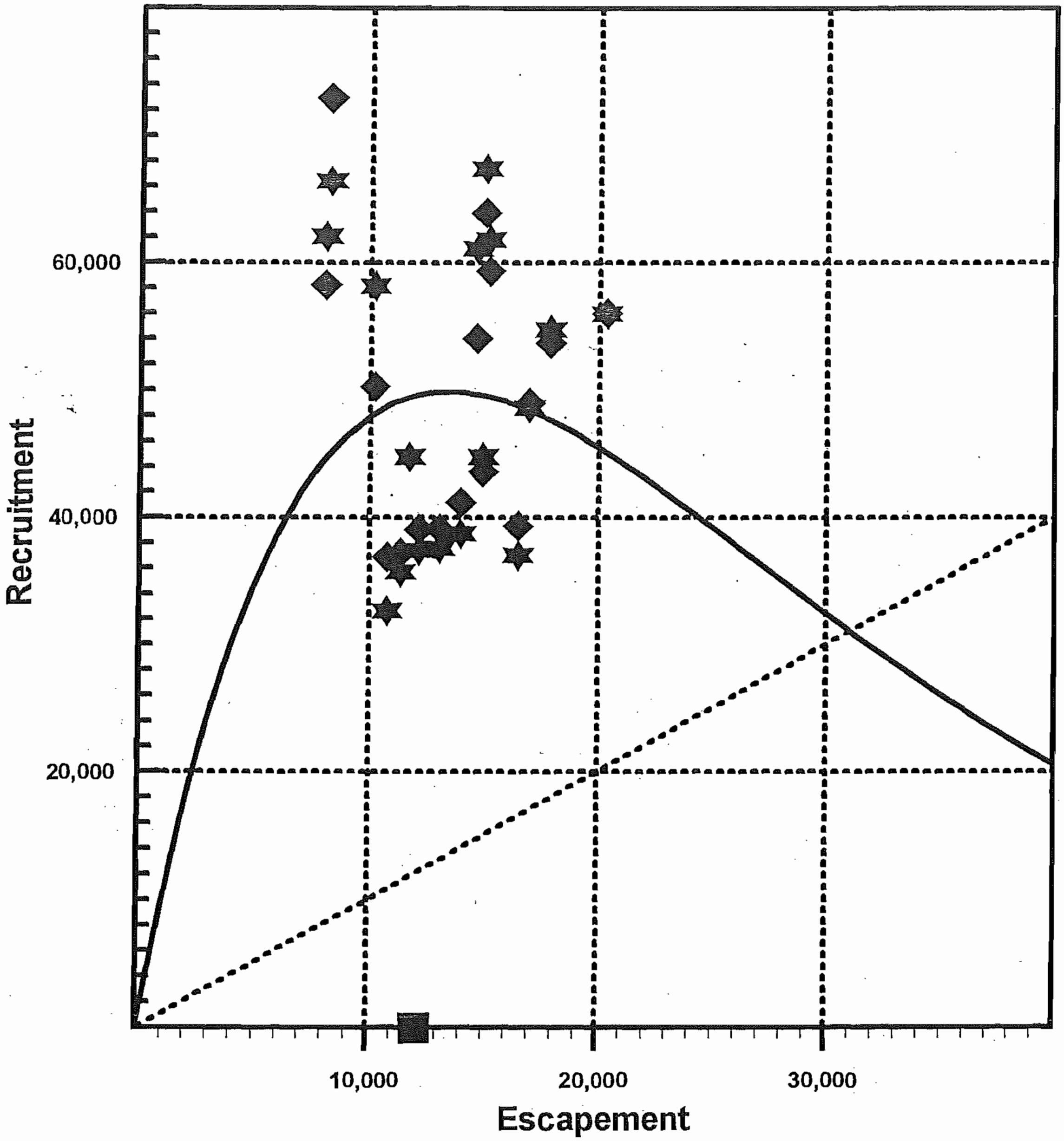


Figure 3.4. Estimated spawner-recruit relationship for the summer stock of chinook salmon spawning in the upper Columbia River with the AR(1) version of Ricker's model. Stars represent individual brood year escapements and resultant recruits as predicted with the CTC chinook model and used as surrogate estimates. Diamonds represent individual brood year escapements and predicted recruits after taking into consideration the auto-regressive Ricker model. The solid square is the estimated MSY escapement point value of 12,143 as measured at Rock Island Dam.

Adjustment to Bonneville Dam: The interim MSY escapement goal estimated for the Columbia upriver summer chinook salmon stock is 12,143 spawners above Rock Island Dam, but the CTC uses escapement at Bonneville in the model. The average inter-dam conversion rate from Bonneville Dam to Rock Island Dam during the years 1973 to 1998 was 0.68 (see Table 3.5 for conversions). Dividing the interim escapement goal of 12,143 spawners by the average rate of 0.68 results in an interim escapement goal at Bonneville Dam of 17,857. The 90% confidence interval range is 13,010 to 23,502 adult chinook salmon from the Columbia upriver summer stock past Bonneville Dam.

Table 3.5. Spawning escapements and subsequent recruits in adult equivalents as predicted by the CTC model, calibration 98-12 for the Columbia upriver summer stock.

Brood Year	Bonneville Escapement	Inter-Dam Conversion Rate	Estimated Spawning Escapement	Estimated Recruits
1979	26,707	0.76	20,297	55,973
1980	23,608	0.62	14,637	61,126
1981	19,185	0.53	10,168	58,183
1982	15,461	0.52	8,040	62,097
1983	15,255	0.54	8,238	66,443
1984	17,621	0.85	14,978	67,450
1985	19,448	0.78	15,169	61,822
1986	21,490	0.83	17,837	54,739
1987	22,531	0.62	13,969	38,784
1988	23,559	0.70	16,491	37,030
1989	23,211	0.73	16,944	48,652
1990	20,447	0.73	14,926	44,801
1991	16,030	0.71	11,381	35,733
1992	14,173	0.76	10,771	32,650
1993	15,366	0.85	13,061	37,609
1994	16,036	0.76	12,187	37,430
1995	15,230	0.77	11,727	44,807

The estimated MSY exploitation rate for the Columbia upriver summer stock needs to be adjusted for the same reason. With the point estimate of MSY escapement of 12,143 fish, the sustained yield is 37,041 chinook salmon. However, the difference between the point value of 12,143 fish and 17,857 (12,143/0.68) or 5,714 fish need to be moved into escapement past Bonneville Dam to account for inter-dam loss. An escapement past Bonneville Dam of 17,857 has an associated sustained yield of 31,327 (37,041-5,714), and thus the associated MSY exploitation rate is estimated at 64% [31,327/(17,857 + 31,327)]. Confidence intervals for the exploitation rate were similarly adjusted.

3.3.4 Discussion

There are several major weaknesses inherent in this stock-production analysis. Contrast in the spawning escapements included in the analysis is low. Escapements ranged from 8,040 in 1982 to 20,297 in 1979, only a 2.5 fold level of variation. Without sufficient contrast, there is a danger that conditional variation in production will obscure effects of spawning abundance. If so, the estimate for S_{MSY} will be misleading.

The issue of measurement error in the data was not addressed in the analysis. Such measurement error arises from substituting estimates of recruitment and especially spawning abundance for actual values when fitting stock-recruit models. Such measurement error obscures the relationship between stock size and recruitment leading to overestimates of the recruitment curve (Hilborn and Walters 1992:288), biasing the estimate of S_{MSY} downward and making the stock appear more productive than it is.

Also, although no measurement error was presented for the data used in this analysis, such measurement error obviously exists. All data used in this analysis are CTC chinook model data rather than actual estimates of annual spawning abundance and resulting recruitments.

The interim MSY escapement goal for the Columbia upriver summer stock of 17,857 (90% confidence interval of 14,203 to 25,038) chinook salmon counted past Bonneville Dam will be used by the CTC until a more thorough analysis based upon detailed stock-production information is completed. To ensure timely completion of a detailed stock-production analysis, the USCTC has supported the funding of research by the Washington Department of Fish and Wildlife to develop a detailed stock-production data base and to identify a biologically based goal for the Columbia River summer stock of chinook salmon. The final report for this study is scheduled for fall of 2000. The CTC will review that report and should be able to adopt an agreed to biological escapement goal for this stock in the spring of 2001.

4. OREGON COASTAL STOCKS

4.1. Summary

Stock-recruit analysis for three Oregon coastal fall chinook stocks, the Nehalem, Siletz, and Siuslaw were conducted. Stock-recruit data bases for these stocks were developed and fit with simple Ricker models to derive scientific estimates of the maximum sustained yield escapement levels (S_{MSY}) and exploitation rates (U_{MSY}) for each stock. Results of this analysis are summarized below:

Stock	Brood Years Analyzed	Adjusted R^2	$\ln\hat{\alpha}$	$\hat{\beta}$	$\hat{S}_{MSY}$ (90% CI)	$\hat{U}_{MSY}$ (90% CI)
Nehalem	1967-1991	0.536	1.878	0.0000977	6,989[a] (5,789 – 9,405)	0.682[b] (0.598 - 0.746)
Siletz	1973-1991	0.795	2.493	0.000273	2,944[a] (2,527 – 3,481)	0.804[b] (0.760 – 0.842)
Siuslaw	1965-1991	0.276	1.577	0.000044	12,925[a] (9,541 - 20,958)	0.573[b] (0.519 – 0.695)

[a] Because this estimate has not been adjusted for process error or measurement error in estimated recruitment or spawning abundance, it is probably biased low ($\hat{S}_{MSY} < S_{MSY}$).

[b] Because this estimate has not been adjusted for process error or measurement error in estimated recruitment or spawning abundance, it is probably biased high ($\hat{U}_{MSY} > U_{MSY}$).

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4.2. Methods

Stock-recruit analysis has been completed for three Oregon coastal fall chinook stocks: the Nehalem, Siletz, and Siuslaw (Figure 4.1). Since the analytic methods used are similar for all three stocks, common methods are explained first, followed by descriptions and analysis for individual stocks.

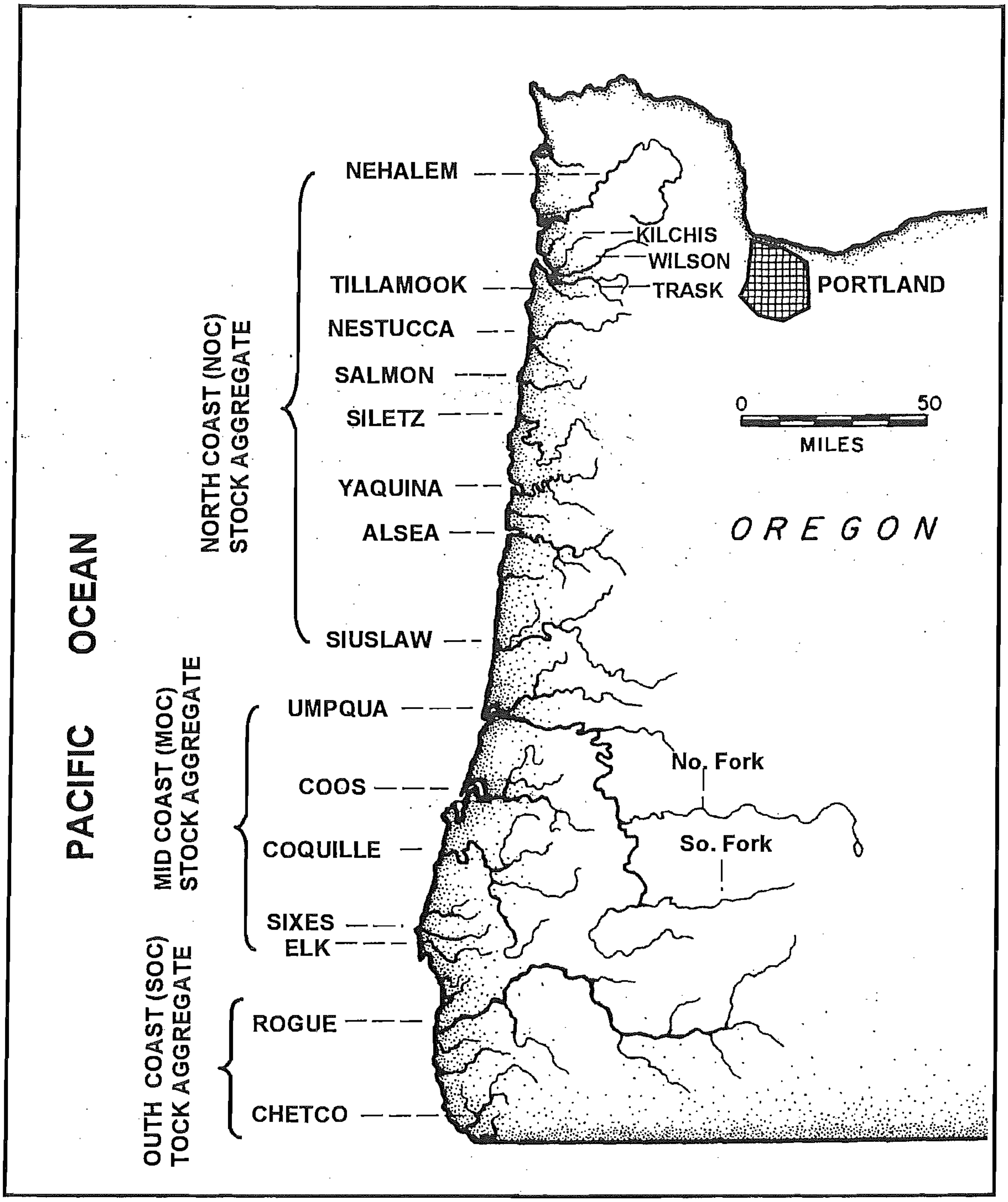

Figure 4.1. Major Oregon coastal river basins that support populations of chinook salmon aggregated by stock groups based on catch distribution.

Spawning Fish Surveys: Oregon Department of Fish and Wildlife (ODFW) conducts a uniform monitoring program for returning chinook salmon in all coastal rivers that produce salmon. The resultant database is similar in character for all coastal rivers. Stock specific variations from this general approach are explained in the stock descriptions. Foot or boat surveys are conducted annually in all three rivers to observe spawning chinook salmon. Surveys began in the early

1950s. Analysis of the three Oregon coastal stocks use spawning survey data from the period 1965 to 1996.

Historical survey sites were not based on a statistical sampling design, but instead, surveys were concentrated in prime chinook spawning habitat. For the years 1967 to 1975 several standard sites in each river basin were annually surveyed. From 1975-1986 usually only one site in each river basin was surveyed. Beginning in 1986 all former standard sites were again surveyed and several new sites were added. Beginning in 1992, many more surveys were conducted. These latter surveys were designed to monitor coho salmon and conducted in areas more attractive to coho than to chinook, however chinook were also counted. Additionally, these more recent surveys were chosen randomly and cover far more of the potential chinook spawning habitat than the standard survey sites. Each of the survey sites is monitored on a regular basis throughout the entire spawning period. A survey is generally repeated every 7-10 days depending upon weather and flow conditions. Counts of live and dead fish are made on each day a survey is made and counted fish are categorized as adults or jacks. Since 1985, scales and lengths have been taken from samples of dead fish and sex noted.

<u>Estimating Escapement:</u> To monitor long term trends in escapement, ODFW uses the density of spawners rather than an estimate of total spawners. The maximum number of live and dead fish (peak count) on any one day among the multiple surveys at each site is used as the annual index. We used the spawner densities in the following manner to estimate total spawner abundance in year y (S_y):

$$\hat{S}_y = \hat{\bar{d}}_y M \hat{p}^{-1} = \frac{\sum_i^n c_i}{\sum_i^n m_i} M \hat{p}^{-1}$$

where: $\bar{d}_y$ = average density of fish per mile in year y;
M = total miles of spawning habitat in the river;
p = probability of fish being observed during the survey;
n = the number of stream segments surveyed in the river;
c_i = peak count of live and dead fish in stream segment i;
m_i = miles surveyed in stream segment i.

The total miles of spawning habitat in each river are based on updated habitat investigations (S. Jacobs, ODFW Corvallis Research Lab, personal communication) and an estimated detectability factor $\hat{p}$ of 0.5 based on an average of estimates from Higley and Williams (1992) and Solazzi (1984). Estimates of average spawner density as reported in ODFW annual survey monitoring reports (Jacobs and Cooney 1997) are thought to be biased high. These surveys have been inconsistently conducted, the index sites were not randomly selected, and were mostly in areas recognized as prime spawning habitat. Adjustments to the survey data for the period 1975-86 have been made previously (Higley and Williams, 1992), and incorporation of the random surveys since 1992 assumes these sites represent sub-optimal spawning sites for chinook and are probably biased low. To estimate the average spawner density for all potential chinook spawning habitat, we believe the mean of the densities from these two survey types is a better estimator:

$$\hat{\bar{d}}_y = \frac{\hat{\bar{d}}'_y + \hat{\bar{d}}''_y}{2},$$

where: $\bar{d}'_y$ is the average density of chinook salmon in the directed survey in year y, and $\bar{d}''_y$ is the average density of chinook salmon in random coho surveys in year y. To adjust historical average densities during years with no information from surveys for coho salmon, a correction factor ϕ was estimated from the most recent six years of data (1992-1997) when both survey methods were employed independently:

$$\hat{\phi} = \frac{1}{2\times 6}\sum_{92}^{97}(1+\frac{\hat{\bar{d}}''_y}{\hat{\bar{d}}'_y}),$$

where: ϕ was estimated as 0.65, 0.70, and 0.78 for the Nehalem, Siletz, and Siuslaw rivers. Total spawner abundance was then used in the stock recruitment analysis. The resulting management parameters from the analysis could be reconverted to spawner density values to correspond with the measurement of spawners from the annual spawning fish monitoring program based on standard index site surveys as follows:

$$\hat{D} = \frac{\hat{Q}}{\hat{\phi}M\hat{p}},$$

where: D= spawner density at the estimated value of a management parameter from the stock-recruitment model; and
Q = a management parameter (S_{MSY}, S_{MSP}, or confidence bounds of these values).

No estimates of variance are available for statistics $\hat{p}$, $\hat{\bar{d}}'_y$, $\hat{\bar{d}}''_y$, $\hat{\phi}$, $\hat{D}$, and subsequently $\hat{S}_y$.

4.2.1. Age Composition of Spawning Escapement

Age composition is required for estimating the different cohorts among the annual terminal runs. Since the early 1980s, scales from chinook carcasses in the three rivers have been collected and analyzed for age composition (Borgerson and Bowden, 1996). However, carcass recoveries are biased by fish size. Large fish are over represented in the recovery samples while small fish are under-represented. Therefore, we estimated age composition as follows: the population of age-2 jacks was estimated directly from peak counts in spawning ground surveys while the proportions of ages 3 and above were estimated by scale data adjusted by information from the mark-recapture study in the Salmon River.

Since 1986 annual mark-recapture studies have been conducted for the exploitation rate indicator stock in the Salmon River. Fall chinook recaptures were stratified by fork length and sex into several categories: <60 cm, 60-79 cm, 80-99 cm, and ≥ 100 cm for males, and <105 cm and ≥ 105 cm for females. A estimated correction factor for each length and sex stratum is:

$$\hat{\gamma}_{l,g} = \frac{1}{Y}\sum_{y=1}^{Y}\frac{\hat{\mathrm{T}}_{y,l,g}}{\tau_{y,l,g}},$$

where: Y = number of years with data;
$T_{y,l,g}$ = abundance in length category l and sex g in year as estimated in a mark-recapture experiment ;
$\tau_{y,l,g}$ = number of fish carcasses recovered in the second event for length category l and sex g in year y.

This estimated correction factor was applied to adult (ages 3 and above) carcasses sampled during spawning ground surveys in the three rivers. The proportion of age a fish in year y was estimated as:

$$\hat{q}_{y,a} = \frac{\sum_{g=1}^{2}\sum_{l=1}^{4}\hat{\gamma}_{l,g}\eta_{y,l,g,a}}{\sum_{a=3}^{6}\sum_{g=1}^{2}\sum_{l=1}^{4}\hat{\gamma}_{l,g}\eta_{y,l,g,a}},$$

where: $\eta_{y,l,g,a}$ = number of chinook carcasses in length category l, sex g, age a recovered in one of the three rivers in year y.

No age data for chinook from these three rivers was collected before 1985. The mean age composition for 1985-1996 was applied to run years before age data were available. Information is available on the sex composition beginning in 1986. However, we did not attempt to stratify estimates on the basis of sex. No estimates of variance for the $\hat{q}_{y,a}$ were calculated.

Freshwater Harvest: Anglers voluntarily report catch information to ODFW on salmon and steelhead tags called punch cards. Data from punch cards are available from 1969 (1967 for the Siuslaw River) to the present and length measurements of some of the samples have been available since 1986. Although anglers are not required to report their catch of jacks, when we examined reported fish length data, it appeared that some jacks had been included. We assume that fish with total length $\leq$ 61 cm (24 inches) were jacks. We first estimated the total jacks included in the punch card reports and subtracted these jacks from the total catch to obtain adult catch. Because no creel surveys have been conducted in these rivers, we used the age compositions from Salmon River creel surveys as a surrogate to obtain freshwater harvest by age in these three rivers, including age-2 jacks. Because there were no creel surveys prior to 1986 in the Salmon River, the average age composition during 1986-96 was used to estimate historical harvest by age.

Ocean Exploitation Rates: The three stocks used in this analysis are thought to have ocean migration and life history patterns similar to the North Oregon Coast exploitation rate indicator stock, reared at the Salmon River hatchery. Therefore, we applied the ocean exploitation rate for Salmon River chinook to the three stocks. The backward cohort method was used to estimate ocean exploitation rates during 1977-1994 brood years when CWT information was available

(CTC 1988). For brood years before 1977 when no CWT data were available, we used the mean estimate of ocean exploitation rate during the base period (brood years 1977-1980) adjusted by estimates of historical fishing effort:

$$\hat{H}_{y,f,a} = \frac{\hat{E}_{y,f,a}}{\hat{E}_{b,f,a}} \hat{H}_{b,f,a},$$

where: $H_{y,f,a}$ = harvest rate in year y fishery f on age a fish;
$E_{y,f,a}$ = fishing effort in year y fishery f on age a fish;
$E_{b,f,a}$ = mean fishing effort during calendar years 1979-1986 (brood year 1977-1980) for fishery f age a fish;
$H_{b,f,a}$ = mean harvest rate during base period (brood years 1977-1980) for fishery f age a fish.

The estimated ocean exploitation rate for year y and age a was the summed estimated ocean harvest rates for all troll, net, and sport fisheries from Alaska to Oregon, i.e., $\hat{H}_{y,a} = \Sigma \hat{H}_{y,f,a}$. No estimates of variance were available for estimated harvest rates, fishing effort targeted on age groups, and subsequently on estimates of ocean exploitation rates.

Recruitment: Recruits were defined as total fish returning to freshwater at the river mouth assuming no ocean harvest:

$$\hat{R}_y = \sum_{a=2}^{6} \left(\frac{\hat{S}_y \hat{q}_{y,a} + \hat{F}_{y,a}}{\prod_{a'=2}^{a} (1 - \hat{H}_{y,a'})} \right),$$

where: $F_{y,a}$ = freshwater sport catch on brood y age a;
$H_{y,a}$ = ocean exploitation rate in total mortality on brood y age a.

No estimates of variance were available for estimated sport catch, estimated exploitation rates, spawning stock size, relative age composition, and subsequently estimates of recruitment.

Stock-Recruit Analysis: Spawners were defined as adult (age 3 and older) fish only, as age 2 jacks were believed to contribute insignificantly to reproduction. However, because jacks contributed significantly to harvest and the terminal run, they were included in the recruitment. Paired spawner and recruit data were fitted with both the Ricker and Beverton-Holt models (Hilborn and Walters 1992). The Ricker Model was selected because it fit the data better than the Beverton-Holt model:

$$\ln(\hat{R}_y / \hat{S}_y) = \ln(\alpha) - \beta \hat{S}_y + r_y,$$

where r_y is the residual (predicted minus actual) recruitment for brood y and is a composite of measurement and process error. Since the recruitment included both adults and jacks while the spawners were adults only, the two variables had different units. As we can estimate the mean ratio θ of jacks to adults in the total production, the number of adult spawners that produce the maximum harvestable surplus (MSY spawners or S_{MSY}) and the exploitation rate at maximum sustainable yield U_{MSY} were obtained by iteratively solving the following equations for $\hat{S}_{MSY}$ and $\hat{U}_{MSY}$:

$$1 = (1 - \hat{\beta} S_{MSY}) \exp(\widehat{\ln \alpha})(1 + \hat{\theta})^{-1} \exp(-\hat{\beta} \hat{S}_{MSY}).$$

The optimal exploitation rate μ_{msy} is:

$$\hat{U}_{msy} = \hat{\beta} \hat{S}_{MSY}.$$

The estimated number of adult spawners that produce maximum production is:

$$\hat{S}_{MSP} = \frac{1}{\hat{\beta}}.$$

<u>Variance and Bias:</u> The original regression provided estimated variances for some estimates of parameters (e.g., $\ln \alpha$ and β) but not others (e.g., S_{MSY}). Therefore, a bootstrap procedure was used to estimate variances for all parameters (Efron and Tibshirani 1993, and McPherson 1990). Residuals r_y were corrected for statistical bias as:

$$r'_y = \frac{\ln(\hat{R}_y) - \ln(R^*_y)}{\sqrt{\left(1 - \frac{2}{n}\right)}},$$

where: R_y^* = predicted recruitment using estimates of $\ln \alpha$ and β from the original regression; n = number of data points (brood years).

Bias corrected residuals were stored, and for each bootstrap iteration b a new set of $\ln(R^+_{y(b)})$ was calculated as $\ln R_y^* + r'_{y(b)}$ where $r'_{y(b)}$ was randomly selected with replacement the old set of residuals. New estimates of parameters were estimated from regression on the pairs $\ln(R^+_{y(b)} / \hat{S}_y)$ and $\hat{S}_y$. One thousand bootstrap samples were so drawn and estimates $\hat{S}_{MSY(b)}$ $\hat{U}_{MSY(b)}$ were calculated for each. Confidence intervals (90%) were estimated with the percentile method by eliminating the lowest and highest 50 samples (90%) of the 1,000 bootstrap distribution as the lower and upper bounds (Efron and Tibshirani 1993). Estimates from the original regression were compared to bootstrap results to evaluate the bias in the original estimates.

Management Parameters: Several management parameters were estimated: spawners needed for maximum sustainable yield (S_{MSY}), spawners needed to attain maximum sustainable recruitment (S_{MSP}), exploitation rate at maximum sustainable yield (U_{MSY}), and exploitation rate at maximum production (U_{MSP}). The 90% confidence intervals for the management parameters are calculated from the bootstrap procedure.

4.2.2. Stock Specific Analysis

Stock 1. Nehalem River Fall Chinook Salmon

Stock and Watershed Description: The Nehalem River is the northernmost large coastal river system in Oregon (Figure 4.1). The river drains an extensive watershed of 667 square miles and includes approximately 782 miles of mainstem and tributaries. The entire watershed lies within a temperate rainforest climate, predominated by Douglas fir forests. A moderate sized estuary provides a productive transition area for juvenile anadromous salmonids. Most of the watershed has been extensively impacted by timber harvest. Residential and commercial development has been minor and concentrated in the estuarine portion of the watershed at the town of Nehalem. Small scale agriculture, primarily livestock and dairy farms, exists in the alluvial floodplains.

There appear to be two discrete runs of chinook to the river. The "summer run," which is the first to enter the river, spawns in the upper river reaches, while the "fall run" begins river entry later and spawns in lower river areas. There is substantial overlap of both spawn timing and location of spawning. The summer run is generally considered to be the smaller of the two runs of chinook in the river. Both races have a predominance of zero age or "ocean type" juvenile rearing pattern. Samples from upstream traps and estuarine seining have observed downstream migrating juveniles in early summer. Studies have shown these fish rear extensively in the estuary prior to entering the ocean during the mid- to late summer of their first year of life (Nicholas and Hankin, 1988). No tagging studies have been conducted with this particular stock, but it is believed they have a far north distribution similar to the North Oregon Coast exploitation rate indicator stock. Tag recoveries of the indicator stock occur predominantly in the SEAK and Northern BC fisheries. Very few recoveries are made in WCVI or Pacific Council Ocean fisheries (Lewis, 1994). A substantial number of fish are also caught in the terminal sport fishery in the estuary and river. Historically, these stocks have not had an escapement goal, rather they were generally monitored to conform to a coastwide comprehensive escapement density target of 60-90 peak fish per mile as measured at standard survey sites.

We estimate that about 121 miles of river contain suitable conditions for chinook to use during spawning. However, chinook are aggregate spawners and tend to concentrate spawning in sections of the river with a large volume of water and gravel combined with optimal flow characteristics. Consequently, we find spawning to be stratified, with high spawner densities in these preferred areas, while some sporadic occurrence of spawners at much lower densities is found throughout the remainder of the 121 miles.

Spawning Surveys: Foot surveys have been conducted for chinook salmon on the river since 1950. There are 6 "standard survey" sites in the database encompassing 5.2 linear miles of prime chinook spawning habitat. During the 1975-85 period only one site (1 mile) was surveyed for 8 years, 2 sites (2 miles) for 1 year, and 3 sites (3 miles) for 1 year. The random surveys began in

1992 and include from 7-18 miles of secondary chinook spawning habitat. Standardization of the spawning fish density index followed the common method previously explained.

Few hatchery fish have been released into the Nehalem River. During the four year period from 1979 to 1981, between 22,000 and 104,000 hatchery chinook were released. Release sites were chosen to be a minimum of 10 miles distance from spawning survey sites. No further releases have occurred. Therefore, the influence of the hatchery fish in the estimates is assumed insignificant.

Age Composition of the Adult Fish Escapement: Age data for the Nehalem River fall chinook have been available annually since 1985. Age composition as determined from scales collected on the spawning grounds were adjusted for size bias by using the mark recapture age composition of the indicator stock at Salmon River. Age 4 and 5 fish are the most predominant age classes in the escapement and on average nearly 52% of adults were age-5 fish (Table 4.1). Age-7 fish were very rare and were combined with age-6 fish. The average age composition was used to apportion the pre- 1985 total escapement estimates into age classes.

Total Estimated Escapement: River basin spawner density indices have varied between 10 and 130 peak fish per mile (mean of 53-peak fish/mile and SD of 28.7 peak fish/mile). The consequent estimated escapement for all ages of chinook for run years 1967-96 ranged from 1,726 to 22,402 fish, with a mean of 9,022 and SD of 4,853 (Table 4.2). For this analysis jacks were not considered viable spawners, but are included as recruitment. Adult spawners (age 3 to 6) varied between 1,597 and 20,341 fish (mean = 8,301, SD = 4,512), a difference of 12.7 times. This broad range of spawning population suggests that statistical stock-recruitment analysis should proceed and it relieves some of the concern of biases in the stock recruitment assessment (Hilborn and Walters, 1992).

Table 4.1. Estimated relative age composition $\hat{q}_{y,a}$ of spawning adult chinook salmon in the Nehalem River.

Return Year	AGE 3	Age 4	Age 5	Age 6+	Sample Size
1985	0.161	0.032	0.791	0.015	63
1986	0.100	0.497	0.341	0.063	62
1987	0.059	0.276	0.598	0.067	92
1988	0.102	0.414	0.428	0.056	244
1989	0.071	0.162	0.717	0.051	224
1990	0.101	0.248	0.523	0.129	136
1991	0.104	0.204	0.506	0.187	70
1992	0.141	0.426	0.379	0.054	285
1993	0.125	0.336	0.521	0.018	234
1994	0.009	0.446	0.517	0.028	108
1995	0.171	0.079	0.706	0.044	91
1996	0.081	0.639	0.191	0.089	438
Mean	0.102	0.313	0.518	0.067	171
SD	0.045	0.180	0.171	0.049	116

Freshwater Harvest: Freshwater and estuarine terminal sport harvest as well as harvest rates shows an increasing trend over the period. Maximum harvest occurred in 1996 with 4,612 landings (Table 4.3). The average harvest rate for the period was 14% (SD= 12%), with a maximum rate of 36% in 1993.

Recruitment: Recruitment is derived from escapement, freshwater catch, and ocean harvest. For the Salmon River indicator stock in brood years 1967 through 1991, the mean ocean harvest rates (including incidental mortality) were 0.033, 0.105, 0.201, 0.390, and 0.420 respectively for age-2 to age-6 fish. These harvest rates were directly applied to Nehalem River fall chinook.

Total recruitment for the 1967-1991 broods have averaged 22,486, ranging from a low of 7,237 for the 1991 brood to 41,586 for the 1980 brood (Table 4.4).

Stock-Recruit Analysis: We examined both Ricker and Beverton-Holt models to fit the spawner recruit data (Hilborn and Walters, 1992). Because the Ricker model resulted in the better fit, we only report results of this model:

$$\ln \hat{\alpha} = 1.88; \ \hat{\beta} = 0.000098$$
$$R^2 = 0.556, \text{ adjusted } R^2 = 0.536, \ n = 25, \ p < 0.001$$

Productivity has varied from 0.8 to 8.1 recruits per spawner for the 1967-1991 broods in the Nehalem River fall chinook stock, with the early 1980s being a highly productive period. The spawner recruit relationship shows a highly productive stock with maximum sustainable yield occurring with 6,989 spawners and maximum sustainable production at 10,240 spawners (Figure 4.2).

Residuals were examined over time and over spawner abundance. No severe problems were observed (Figure 4.3). Residuals were also examined for autocorrelation. The lag $k = 1$ to 12 autocorrelation function ranged from –0.286 to 0.199, and the P values were 0.291 to 0.863 (Figure 4.4).

Variance and Statistical Bias: The mean bootstrap estimate of S_{MSY} was 7,215 (Table 4.5), 226 fish more than the regression estimate of 6,989, indicating only a small statistical bias of 3.2%. The estimated SE for S_{MSY} from the bootstrap is 1,231 fish, representing a coefficient of variance of 17%. The mean S_{MSP} from the bootstrap was 10,779, which was 539 fish more than the regression estimate, indicating a statistical bias of 5.3% in the regression estimate. The coefficient of variation for S_{MSP} was 24%. This indicates the model estimates have relatively minor statistical bias.

Table 4.2. Estimated escapement $\hat{S}_y$ and estimated escapement by age ($\hat{S}_y \hat{q}_{y,a}$) of chinook salmon in the Nehalem River from 1967 to 1996. Estimates from carcasses recovered from the spawning grounds were corrected with information from Salmon River mark-recapture studies. Estimated mean relative age compositions $\hat{q}_a$ for 1986-1996 were applied to run years 1967-1985.

	Estimated Spawner Density Run (fish/Mi.)		Estimated Escapement by Age					Estimate Escapemen	
Year	Adul	Jac	Age	Age	Age	Age	Age	Adul	Total
196	29.	1.	25	46	1,43	2,37	30	4,58	4,84
196	19.	1.	25	30	94	1,57	20	3,03	3,28
196	10.	0.	12	16	50	82	10	1,59	1,72
197	30.	1.	23	49	1,52	2,51	32	4,85	5,09
197	37.	3.	51	60	1,85	3,07	39	5,93	6,45
197	24.	1	2,19	39	1,20	1,99	25	3,84	6,04
197	50.	9.	1,46	80	2,46	4,07	52	7,86	9,32
197	41.	4.	73	66	2,05	3,39	43	6,55	7,28
197	33.	11.	1,81	53	1,62	2,69	34	5,19	7,01
197	62.	16.	2,59	99	3,07	5,08	65	9,80	12,40
197	73.	5.	79	1,17	3,59	5,94	76	11,47	12,27
197	76.	2.	39	1,22	3,77	6,24	80	12,05	12,45
197	77.	0.	13	1,24	3,82	6,32	81	12,20	12,33
198	35.	0.	14	56	1,73	2,87	37	5,55	5,70
198	68.	0.	6	1,09	3,36	5,57	71	10,75	10,81
198	32.	5.	86	51	1,59	2,63	34	5,08	5,95
198	28.	1.	26	45	1,38	2,29	29	4,43	4,69
198	129.	13.	2,06	2,07	6,96	10,54	1,35	20,34	22,40
198	118.	16.	2,66	1,90	5,84	9,67	1,24	18,67	21,33
198	66.	4.	75	1,03	5,15	3,54	65	10,38	11,14
198	86.	3.	51	79	3,73	8,11	91	13,56	14,07
198	94.	2.	39	1,51	6,15	6,37	83	14,88	15,28
198	66.	1.	30	73	1,68	7,44	52	10,38	10,69
199	32.	1.	30	51	1,26	2,66	65	5,10	5,40
199	35.	1.	27	57	1,13	2,81	1,03	5,55	5,82
199	57.	1.	30	1,27	3,86	3,43	49	9,06	9,36
199	34.	1.	21	66	1,79	2,78	9	5,34	5,55
199	41.	1.	26	6	2,89	3,35	17	6,48	6,75
199	33.	3.	51	88	41	3,67	22	5,19	5,70
199	58.	1.	18	74	5,88	1,75	82	9,21	9,39

Table 4.3. Estimated freshwater harvest by age and overall estimates of freshwater harvest rates for Nehalem fall chinook salmon, 1969-1996. Estimated relative age composition is adopted from Salmon River creel census. Estimated mean age composition between 1986-1996 is applied to run years 1969-1985.

Run Year	Age 2	Age 3	Age 4	Age 5	Age 6	Estimated Total Catch	Estimated Harvest Rate
1969	87	122	186	190	21	605	0.250
1970	45	63	96	98	11	313	0.060
1971	79	111	168	172	19	549	0.080
1972	47	66	100	102	11	326	0.071
1973	31	43	66	67	7	215	0.024
1974	28	39	59	61	7	193	0.026
1975	16	23	35	36	4	114	0.016
1976	16	22	34	34	4	110	0.011
1977	95	133	203	207	23	661	0.045
1978	140	197	300	307	34	978	0.072
1979	99	139	212	217	24	690	0.051
1980	128	180	274	281	31	894	0.077
1981	81	114	173	177	19	564	0.046
1982	92	129	196	201	22	640	0.094
1983	72	101	153	157	17	500	0.071
1984	200	280	427	437	48	1,391	0.061
1985	125	176	267	273	30	871	0.040
1986	408	325	369	581	73	1,756	0.138
1987	116	569	582	822	87	2,176	0.136
1988	164	40	1,417	535	90	2,245	0.129
1989	1,094	343	129	976	32	2,573	0.197
1990	262	694	567	213	64	1,800	0.252
1991	338	189	1,007	744	11	2,289	0.274
1992	490	361	670	1,220	241	2,982	0.209
1993	118	1,327	1,546	791	118	3,901	0.364
1994	524	122	1,610	881	46	3,183	0.306
1995	373	872	298	2,062	96	3,700	0.345
1996	196	2,069	1,575	511	261	4,612	0.281

Table 4.4. Estimated recruitment by age $\hat{R}_{y,a}$ and estimated spawners $\hat{S}_y$ for the Nehalem River fall chinook stock, brood years 1967-1991.

Brood Year	Spawners $\hat{S}_y$	Recruits ($\hat{R}_{y,a}$) if no ocean fishery impacts: Age 2	Age 3	Age 4	Age 5	Age 6	$\hat{R}_y$	$\hat{R}_y/\hat{S}_y$	$\ln(\hat{R}_y/\hat{S}_y$
1967	4,587	223	644	2,871	4,555	1,607	9,900	2.2	0.77
1968	3,030	289	825	1,846	8,713	1,307	12,980	4.3	1.45
1969	1,597	614	525	3,501	7,297	1,026	12,963	8.1	2.09
1970	4,857	2,321	965	2,944	5,269	1,758	13,256	2.7	1.00
1971	5,938	1,543	805	2,246	10,276	2,222	17,092	2.9	1.06
1972	3,849	784	624	4,219	14,153	3,323	23,103	6.0	1.79
1973	7,863	1,881	1,161	5,435	17,896	3,818	30,191	3.8	1.35
1974	6,552	2,687	1,501	6,144	17,655	2,289	30,276	4.6	1.53
1975	5,197	922	1,668	6,107	8,738	3,706	21,142	4.1	1.40
1976	9,807	559	1,617	3,181	16,111	1,680	23,147	2.4	0.86
1977	11,478	243	941	6,707	7,488	1,476	16,855	1.5	0.38
1978	12,059	288	1,637	3,711	11,033	11,290	27,959	2.3	0.84
1979	12,205	153	753	2,250	28,497	5,918	37,572	3.1	1.12
1980	5,555	990	654	9,857	26,599	3,485	41,586	7.5	2.01
1981	10,752	348	2,793	9,090	10,660	5,505	28,396	2.6	0.97
1982	5,085	2,319	2,474	8,429	22,440	3,067	38,727	7.6	2.03
1983	4,431	2,835	1,465	5,325	11,885	3,276	24,786	5.6	1.72
1984	20,341	1,199	1,517	9,920	18,436	2,185	33,257	1.6	0.49
1985	18,670	656	1,731	2,276	5,985	5,028	15,677	0.8	-0.17
1986	10,389	575	1,190	2,366	8,759	4,375	17,265	1.7	0.51
1987	13,560	1,448	1,388	2,975	9,342	1,221	16,374	1.2	0.19
1988	14,889	587	874	6,754	10,786	957	19,959	1.3	0.29
1989	10,389	638	1,855	4,839	10,140	1,058	18,530	1.8	0.58
1990	5,104	811	2,251	6,244	10,109	4,499	23,914	4.7	1.54
1991	5,557	341	209	1,024	5,390	273	7,237	1.3	0.26
Mean	8,550	1,010	1,283	4,810	12,329	3,054	22,486	3.4	1.04
St. Dev.	4,855	811	639	2,596	6,447	2,309	9,170	2.1	0.64

Management parameters: Spawners needed for maximum sustainable yield, and maximum production were accepted as estimated from the Ricker model. Estimated MSY occurred at about 7,000 spawners (90% CI = 5,789 to 9,405), and estimated MSP occurred at about 10,000 spawners (90% CI= 7,889 to 15,279). The theoretical exploitation rates at MSY are an estimated 68% and at MSP 56%.

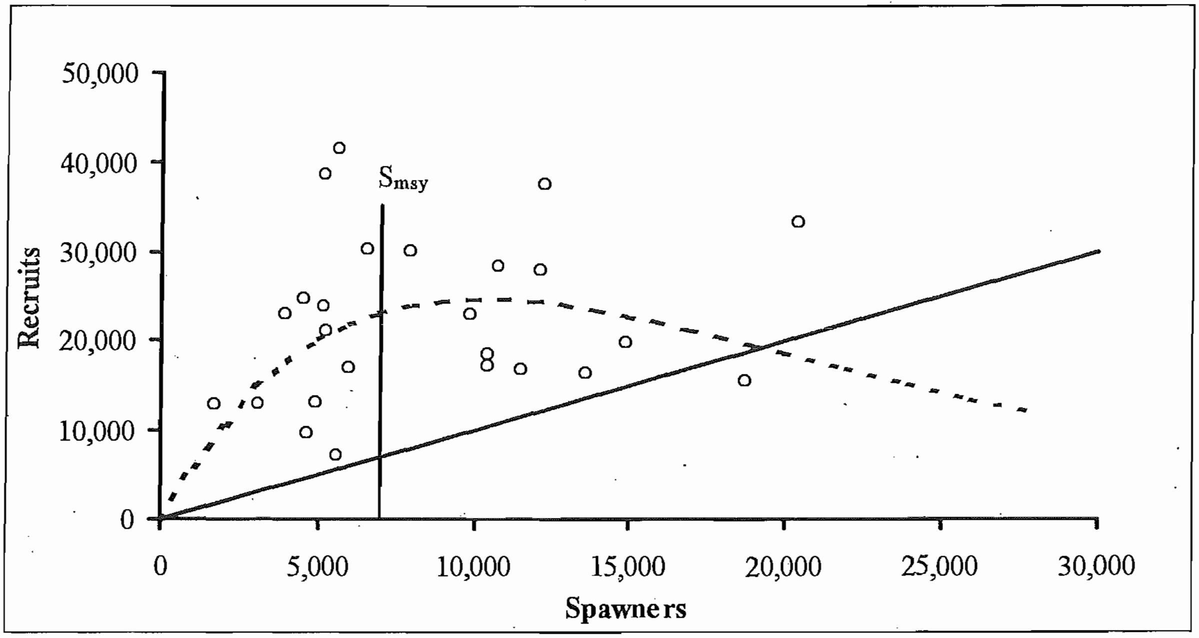

Figure 4.2. Estimated relationship between spawners and recruits for Nehalem River fall chinook, brood years 1967-1991 as modeled by the Ricker function.

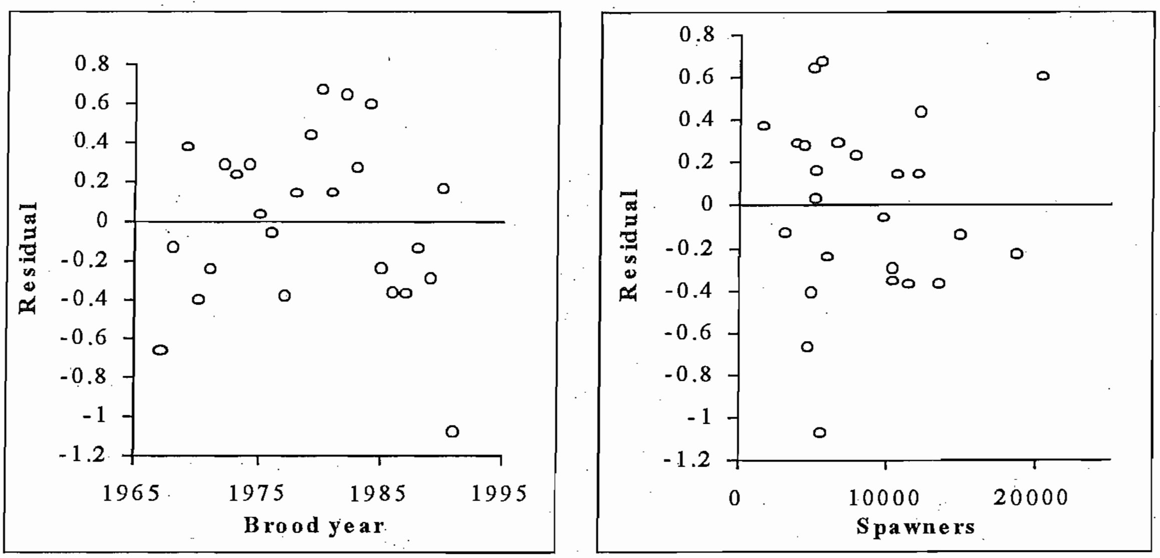

Figure 4.3. Residuals over time and over the number of estimated spawners for the Nehalem River fall chinook salmon.

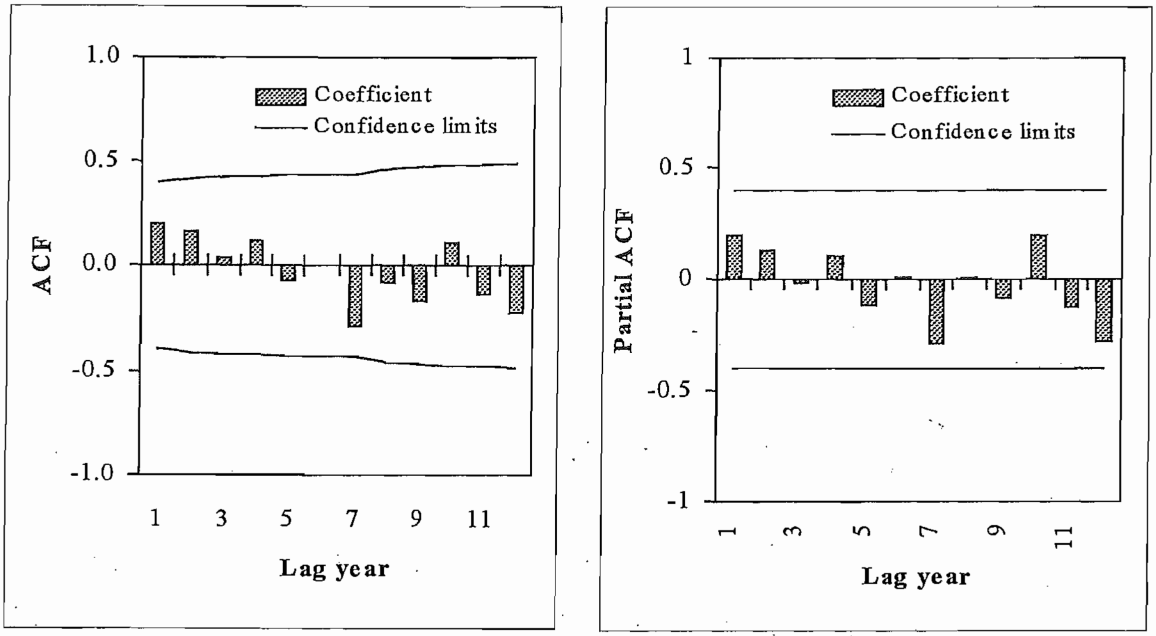

Figure 4.4. Autocorrelation and partial autocorrelation among residuals from the fit of Ricker's model to estimates of spawners and recruitment for the Nehalem River fall chinook salmon stock.

Table 4.5. Parameter estimates from the fit of Ricker's model to estimates of spawners and recruitment for the Nehalem River fall chinook salmon stock.

	$\hat{\ln\alpha}$	$\hat{\beta}$	$\hat{S}_{MSY}$	$\hat{U}_{MSY}$	$\hat{S}_{MSP}$	$\hat{U}_{MSP}$
Model Estimate	1.878	0.0000977	6,989	0.682	10,240	0.564
Bootstrap Mean	1.874	0.000097	7,215	0.679	10,779	0.554
SE	0.185	0.000019	1,231	0.0453	2,596	0.084
CV	9.8%	19.6%	17%	6.7%	24%	15%
Lower 90% CI	1,559	1.27E-04	5,789	0.598	7,889	0.400
Upper 90% CI	2.165	6.55E-05	9,405	0.746	15,279	0.673

Stock Specific Discussion: The accuracy and precision of escapement data has the most significant influence on stock recruitment analysis because not only are the spawners but also the recruits derived from escapement. There are several potential problems associated with these escapement data as used in our abundance estimation procedure and are discussed in the general discussion section of this chapter, as these problems are common to all three stocks analyzed in this section.

Index survey sites were not chosen at random in the Nehalem River but are suspected to be biased high with regard to the total identified spawning habitat of 120.8 miles. Prior to 1975 the agency surveyed 6 sites (5.2 miles). For the period 1975-1986 the agency reduced surveys to only one index site (1 mile), which happened to be the historically highest density spawning site. Consequently the estimates of spawning chinook in the 120.8 miles for this time period were based entirely on counts of fish in one mile of survey. Since 1986, the six standard sites have been annually surveyed, and in 1992 additional surveys were conducted randomly in habitat more suitable for coho spawning but also utilized by some chinook. Standardizing all these discontinuous sets of data to derive a uniform set of escapement estimates to analyze, no doubt added a lot of measurement error, which is inherent in the analysis.

Age composition data is only available back to 1985 for the Nehalem river chinook stock, and the mean ages were used to estimate the cohorts for brood years prior to 1986. This causes a distinct problem with the estimation of recruits for earlier years. This problem is prevalent in all three stocks and is discussed in detail in the general discussion section of the chapter.

For the period analyzed, 53% of the years (16 years) had escapements below the MSY point estimate, while 47% (24 years) were above the MSY point estimate (Figure 4.5). The mid-1980s appeared to be a very productive period and the largest escapements occurred during this time period. This is typical of many of the Oregon Coastal stocks as well as the Upriver Bright chinook stock in the Columbia River (CTC, 1999). It appears that marine conditions were optimal for many stocks that rear in the northeastern Pacific during that period. However, we see the Nehalem chinook escapements since 1990 have returned to about the escapement goal level, as is typical of many of the other regional stocks.

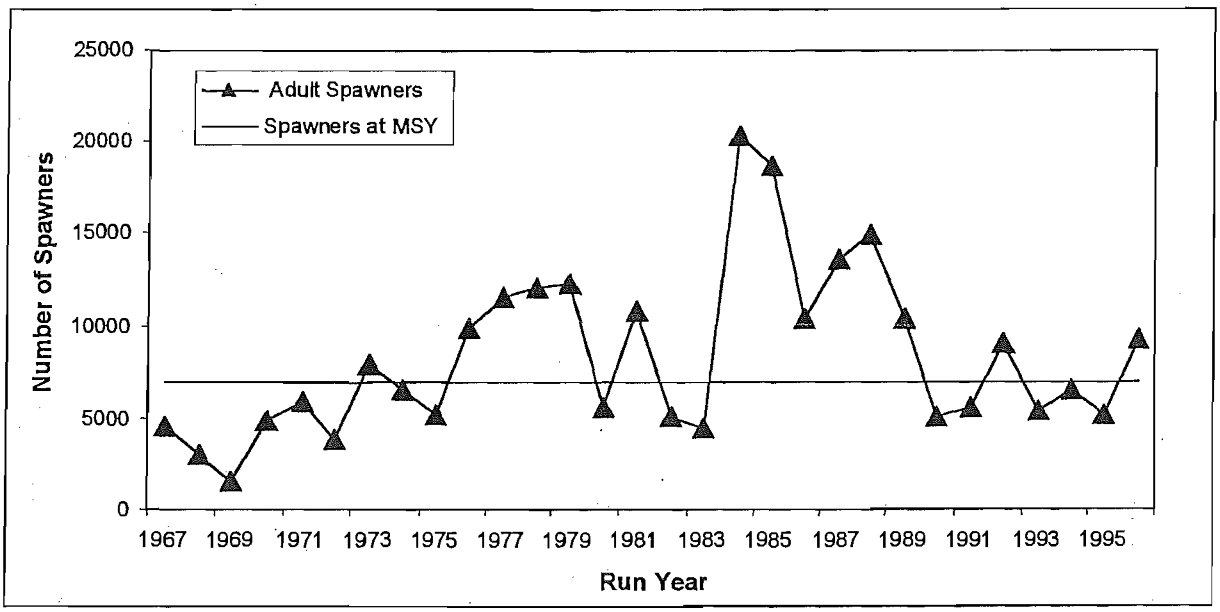

Figure 4.5. Comparison of the annual estimated density of adult chinook spawners with the point estimate of spawners needed for maximum sustainable yield, Nehalem River, 1967-1996.

Stock 2. Siletz River Fall Chinook Salmon

X

Stock and Watershed Description: The Siletz River is a moderate sized coastal river system draining the coast range mountains along the central Oregon coast (Figure 4.1). The river drains a forested watershed of 202 square miles and includes approximately 390 miles of main-stem and tributaries. The entire watershed lies within a temperate rainforest climate, predominated by Douglas fir forests. A small estuary provides a productive transition area for juvenile anadromous salmonids. Most of the watershed has been severely impacted by timber harvest. Residential development has been minor along the banks of the river and at the town of Siletz located at river mile 25. More intense development at Lincoln City is located at the north shore of the estuary. Small scale agriculture, primarily livestock and dairy farms, exists along some of the riparian areas, near the small town of Siletz.

There are two discrete runs of mature chinook salmon in the Siletz River. The "spring run" enters the river beginning in late May, migrating to the upper river area. The fall run enters the river in late summer and fall and spawns in lower river areas. There is substantial overlap of both spawn timing and location of spawning. The spring run is much smaller than the fall run of chinook. Nothing is known of the juvenile life history of the spring run fish; however, the fall run fish have a predominance of zero age or "ocean type" juvenile rearing pattern. Juveniles sampled in the river from June to October were all under-yearlings, and scale patterns on returning adults showed that 99% migrated to the ocean as under-yearlings. No tagging studies have been made with this particular stock, but it is believed they have a far north distribution similar to the North Oregon Coast exploitation rate indicator stock. Tag recoveries of the indicator stock occur predominantly in the SEAK and Northern BC fisheries. Very few recoveries are made in WCVI or Pacific Council Ocean fisheries. A substantial number of fish are also caught in the terminal sport fishery in the estuary and river. This stock has not had an escapement goal; rather they were generally monitored to conform to a coastwide comprehensive escapement target density of 60-90 peak fish per mile as measured at standard survey sites. Historically, hatchery releases have had little influence on the wild fall chinook population in the Siletz River. Only 45,000 hatchery fall chinook fingerlings for brood years 1967 and 1968 were released (Nicholas and Hankin, 1988, and Jacobs and Cooney, 1997).

Spawning Surveys: Foot and boat surveys have been conducted for chinook salmon on the Siletz River since 1952. There are four standard survey sites encompassing 4.7 linear miles of prime chinook spawning habitat. During the 1975-85 period only one site (1.2 miles) was surveyed for eight years, and two sites (2.2 miles) for three years. The random surveys began in 1992 and include from 11-17 miles of secondary chinook spawning habitat. Standardization of the spawning fish density index (d_y) followed the common method as explained in the Oregon methods section. A recent study indicates that the total spawning habitat M, is 98.5 miles in the Siletz River (S. Jacobs, ODFW Corvallis Research Lab, personal communication). Because very few hatchery fingerlings and no smolts have been released in the Siletz River and the releases occurred in areas that were not surveyed, hatchery influence was not considered in the estimates.

Table 4.6. Estimated age compositions of Siletz River fall chinook salmon. The data from carcasses recovered from the spawning grounds were corrected by Salmon River mark-recapture studies. Age-2 jacks were excluded.

Return Year	Age 3	Age 4	Age 5	Age 6+	Sample Size
1986	0.437	0.152	0.325	0.085	238
1987	0.051	0.287	0.469	0.193	185
1988	0.120	0.375	0.427	0.078	85
1989	0.018	0.289	0.640	0.053	71
1990	0.448	0.160	0.258	0.133	56
1991	0.054	0.533	0.361	0.052	196
1992	0.150	0.179	0.510	0.161	256
1993	0.032	0.494	0.408	0.065	129
1994	0.033	0.251	0.653	0.064	236
1995	0.080	0.289	0.539	0.091	163
1996	0.053	0.130	0.730	0.086	251
Mean	0.134	0.285	0.484	0.097	170
SD	0.157	0.135	0.148	0.046	75

Age Composition of the Adult Fish Escapement: Age data from scale analysis for Siletz adult fall chinook spawners have been available since 1986 (Borgerson and Bowden, 1996). Age composition, determined from scales collected on the spawning grounds, were adjusted for size bias by using the mark recapture age composition of the indicator stock at Salmon River. Age-4 and age-5 fish dominate the escapement and on average an estimated 48% of adults were age-5 fish (Table 4.6). Age-7 fish were very rare and were combined with age 6 fish for our analysis. The average estimated age composition was used to apportion the pre-1986 total escapement estimates into age classes.

Total Estimated Escapement: River basin spawner density indices have varied between 8 and 77 peak fish per mile. The consequent estimated escapement for all ages of chinook for run years 1967-1996 ranged from 1,080 to 10,680 fish, with a mean of 4,525 and SD of 2,213 (Table 4.7). On average age-2 jacks comprised an estimated 10% of total escapement (SD = 11%, n = 30), however, jacks are not considered viable spawners, but are included as recruitment. Estimated abundance of spawners (age 3 to 6) varied between 780 and 10,475 fish (mean = 4,192 and SD = 2,254), a difference of 13.4 times. This broad range of spawning population suggests that statistical stock-recruitment analysis should proceed and it relieves some of the concern of biases in the stock recruitment assessment (Hilborn and Walters, 1992).

Freshwater Harvest: Estimates of freshwater and estuarine terminal sport harvest shows an increasing trend over the period, however estimated harvest rates have not shown a similar trend.

Table 4.7. Estimated escapement by age of fall chinook in the Siletz River from 1967 to 1996. Estimated relative age compositions from carcasses recovered from the spawning grounds were corrected with information from Salmon River mark-recapture studies. Estimated relative age compositions averaged over 1986-1996 were applied to run years 1967-1985.

	Estimated Spawner Density Run (fish/mi.)			Estimated Escapement by Age					Estimated Escapement	
Year	Adult	Jack	Age	Age	Age	Age	Age	Adul	Tota
1967	19.7	2.6	359	364	774	1,312	262	2,712	3,070
1968	8.9	2.8	380	166	352	596	119	1,233	1,613
1969	5.7	2.2	300	105	223	377	75	780	1,080
1970	27.6	4.8	665	512	1,088	1,843	368	3,810	4,475
1971	18.5	2.9	402	342	727	1,232	246	2,547	2,949
1972	17.9	9.9	1,360	331	704	1,192	238	2,464	3,825
1973	21.9	0.2	29	405	862	1,460	291	3,019	3,048
1974	18.6	1.0	131	344	732	1,240	248	2,564	2,696
1975	15.0	5.7	786	277	589	997	199	2,062	2,848
1976	9.6	6.8	944	178	378	641	128	1,326	2,269
1977	24.0	1.1	157	445	946	1,603	320	3,314	3,471
1978	15.0	0.0	0	277	589	997	199	2,062	2,062
1979	52.3	4.0	550	969	2,060	3,491	697	7,217	7,768
1980	26.7	2.1	286	494	1,051	1,780	355	3,680	3,967
1981	32.2	0.9	120	596	1,266	2,145	428	4,435	4,555
1982	24.8	0.7	99	459	975	1,652	330	3,415	3,514
1983	15.5	1.7	236	287	610	1,033	206	2,136	2,372
1984	25.1	1.7	236	465	988	1,674	334	3,461	3,697
1985	48.1	1.7	236	890	1,892	3,206	640	6,628	6,864
1986	48.9	6.2	851	2,952	1,026	2,195	576	6,748	7,599
1987	33.2	1.1	147	231	1,314	2,148	883	4,577	4,724
1988	56.6	0.9	117	937	2,924	3,336	608	7,805	7,922
1989	31.9	2.1	293	79	1,272	2,816	233	4,401	4,694
1990	31.3	1.7	235	1,934	692	1,112	575	4,313	4,548
1991	40.9	3.6	499	301	3,004	2,033	294	5,633	6,132
1992	43.8	0.9	117	909	1,083	3,080	973	6,044	6,161
1993	31.5	0.6	88	140	2,146	1,774	282	4,342	4,430
1994	76.0	1.5	205	346	2,628	6,835	665	10,475	10,680
1995	37.4	0.6	88	414	1,494	2,784	472	5,164	5,252
1996	53.6	0.4	59	395	963	5,398	639	7,394	7,452

Table 4.8. Estimated freshwater harvest by age and estimated freshwater harvest rates for Siletz River fall chinook, 1969-1996. Estimated age compositions are adopted from Salmon River creel census. Estimated relative age composition averaged over 1986-1996 is applied for run years 1969-1985.

Run Year	Age 2	Age 3	Age 4	Age 5	Age 6	Estimated Total Catch	Estimated Harvest Rate
1969	135	190	289	295	32	941	0.466
1970	77	109	165	169	19	539	0.108
1971	94	132	201	206	23	656	0.182
1972	79	111	169	173	19	551	0.126
1973	59	83	127	130	14	414	0.120
1974	43	60	91	94	10	298	0.100
1975	100	141	214	219	24	697	0.197
1976	113	159	242	248	27	790	0.258
1977	97	136	206	211	23	673	0.162
1978	262	368	560	573	63	1,825	0.470
1979	270	379	576	590	65	1,880	0.195
1980	183	256	390	399	44	1,271	0.243
1981	371	520	791	810	89	2,580	0.362
1982	176	246	375	384	42	1,222	0.258
1983	162	228	346	355	39	1,130	0.323
1984	177	249	379	388	42	1,235	0.250
1985	169	237	360	369	40	1,176	0.146
1986	302	240	273	430	54	1,299	0.146
1987	76	373	382	539	57	1,427	0.232
1988	111	27	957	361	61	1,516	0.161
1989	1,092	343	128	974	32	2,569	0.354
1990	173	460	375	141	42	1,191	0.208
1991	215	120	642	474	7	1,459	0.192
1992	255	188	348	634	125	1,550	0.201
1993	77	860	1,002	513	77	2,528	0.363
1994	358	83	1,101	602	31	2,176	0.169
1995	349	816	278	1,930	90	3,463	0.397
1996	139	1,474	1,122	364	186	3,283	0.306

Maximum harvest occurred in 1995 with an estimated 3,463 landings (Table 4.8). The average estimated harvest rate for the period was 24% (SD=10%), with a maximum rate of 47% estimated for both 1969 and 1978.

Recruitment: Recruitment defined as total returns to the freshwater assuming no ocean fishery impacts, generally increased over the time period (Table 4.9). For brood years 1967-1972 estimated recruits per brood were less than 7,000. Since 1973 estimated recruits varied between 10,000 to 20,000 per brood except for brood year 1985, which had an estimate of 7,110, and brood year 1989 that produced an estimated 26,206 recruits. For the 25 broods from 1967-1991, the estimated ratio of total returns to spawners ($\hat{R}_y/\hat{S}_y$) varied between 1.1 and 10.7 (mean = 4.3, SD = 2.3).

Table 4.9. Estimated recruitment by age $\hat{R}_{y,a}$ and estimated spawners $\hat{S}_y$ for the Siletz River fall chinook, brood years 1967-1991.

Brood Year	Spawners $\hat{S}_y$	Recruits ($\hat{R}_{y,a}$) if no ocean fishery impacts: Age 2	Age 3	Age 4	Age 5	Age 6	$\hat{R}_y$	$\hat{R}_y/\hat{S}_y$	$\ln(\hat{R}_y/\hat{S}_y$
1967	2,712	449	716	1,315	2,965	922	6,366	2.3	0.853
1968	1,233	769	547	1,234	3,345	758	6,653	5.4	1.685
1969	780	511	506	1,369	2,816	652	5,855	7.5	2.016
1970	3,810	1,487	559	1,148	2,348	414	5,956	1.6	0.447
1971	2,547	91	461	1,084	1,785	966	4,388	1.7	0.544
1972	2,464	180	471	843	4,171	1,037	6,702	2.7	1.001
1973	3,019	910	383	1,650	4,287	3,465	10,694	3.5	1.265
1974	2,564	1,089	669	1,731	11,015	2,273	16,777	6.5	1.878
1975	2,062	262	754	3,993	6,028	2,597	13,635	6.6	1.889
1976	1,326	271	1,576	2,276	8,282	1,727	14,132	10.7	2.367
1977	3,314	859	947	3,898	5,375	1,155	12,233	3.7	1.306
1978	2,062	492	1,510	2,801	6,240	3,023	14,066	6.8	1.920
1979	7,217	510	821	1,396	5,352	3,153	11,231	1.6	0.442
1980	3,680	285	609	1,982	9,558	3,005	15,439	4.2	1.434
1981	4,435	410	847	3,350	6,788	5,188	16,584	3.7	1.319
1982	3,415	424	1,342	1,981	6,750	2,217	12,714	3.7	1.314
1983	2,136	412	3,442	2,091	6,357	1,559	13,860	6.5	1.870
1984	3,461	1,189	671	5,081	8,298	1,870	17,109	4.9	1.598
1985	6,628	232	1,072	1,758	2,603	1,445	7,110	1.1	0.070
1986	6,748	236	466	1,379	6,181	6,561	14,822	2.2	0.787
1987	4,577	1,437	2,749	5,070	7,456	2,027	18,739	4.1	1.410
1988	7,805	425	482	2,133	6,896	2,969	12,906	1.7	0.501
1989	4,401	747	1,244	4,561	17,814	1,840	26,206	6.0	1.784
1990	4,313	381	1,129	5,168	8,314	3,419	18,411	4.3	1.451
1991	5,633	170	495	2,564	13,684	2,322	19,235	3.4	1.228

<u>Stock-Recruit Analysis:</u> We examined both Ricker and Beverton-Holt models to fit the spawner-recruit data (Hilborn and Walters, 1992). Because the Ricker model resulted in a better fit, we adopted this model (Figure 4.6; BY 1967-1991):

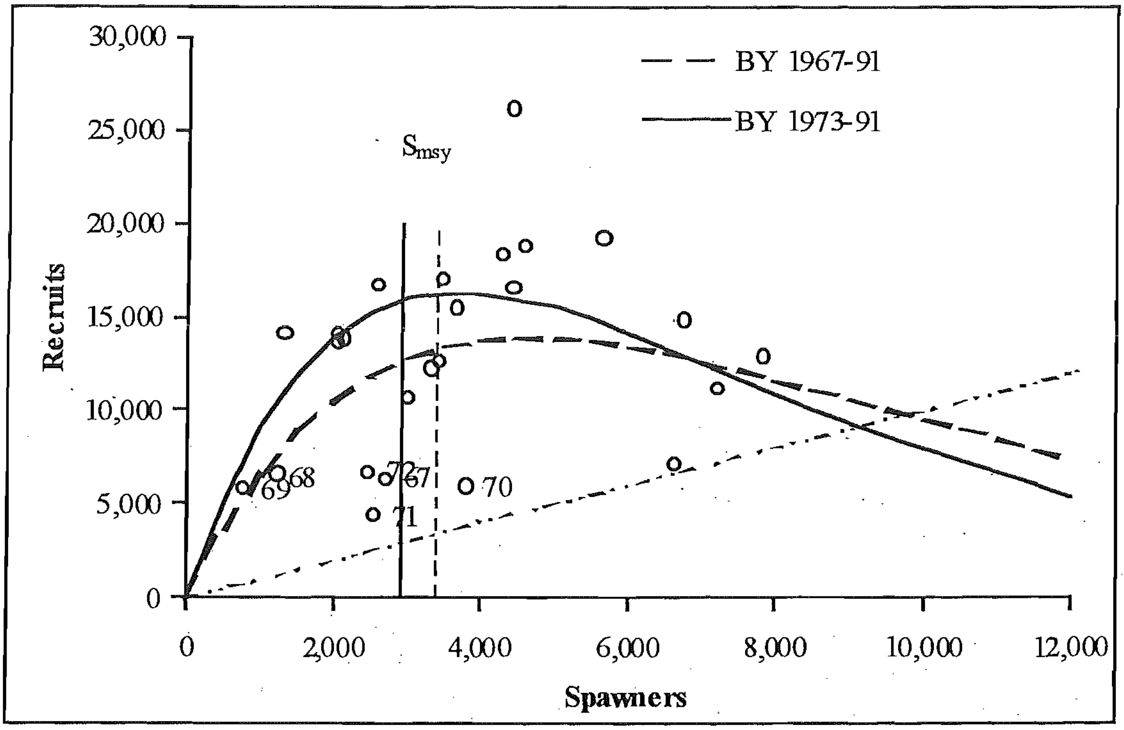

Figure 4.6. Estimated relationship between spawners and recruits for Siletz River fall chinook in two time periods, brood years 1967-1991 (25 years) and brood years 1973-1991 (19 years).

$$\ln \hat{\alpha} = 2.09; \ \hat{\beta} = 0.000214$$
$$R^2 = 0.480, \text{ adjusted } R^2 = 0.457, n = 25, P < 0.001.$$

However, when we examined the residuals over time and spawner abundance, plots showed a clear temporal pattern (Figure 4.7). The residuals for brood years 1967-1972 appeared biased low. Also, significant autocorrelation and partial autocorrelation were detected for lag $k = 1$ (P = 0.029) (Figure 4.8). These plots indicated that there were auto-regressive and nonstationary problems, which were time-series biases, in this stock-recruitment database. The increasing recruitment over brood years might have resulted from major regime shifts in natural environment (T. Nickelson, ODFW Corvallis, personal communication). This nonstationary process is difficult to correct. Because no statistical analysis of nonstationary data on spawning abundance and production can result in an accurate estimate of optimal production for the future (D. Bernard, ADF&G, Anchorage, personal communication), we decided to remove brood years 1967-1972 and re-analyze the stock-recruit relationship.

The resultant model for brood years 1973-1991 shows an improved regression (Figure 4.6; BY 1973-91):

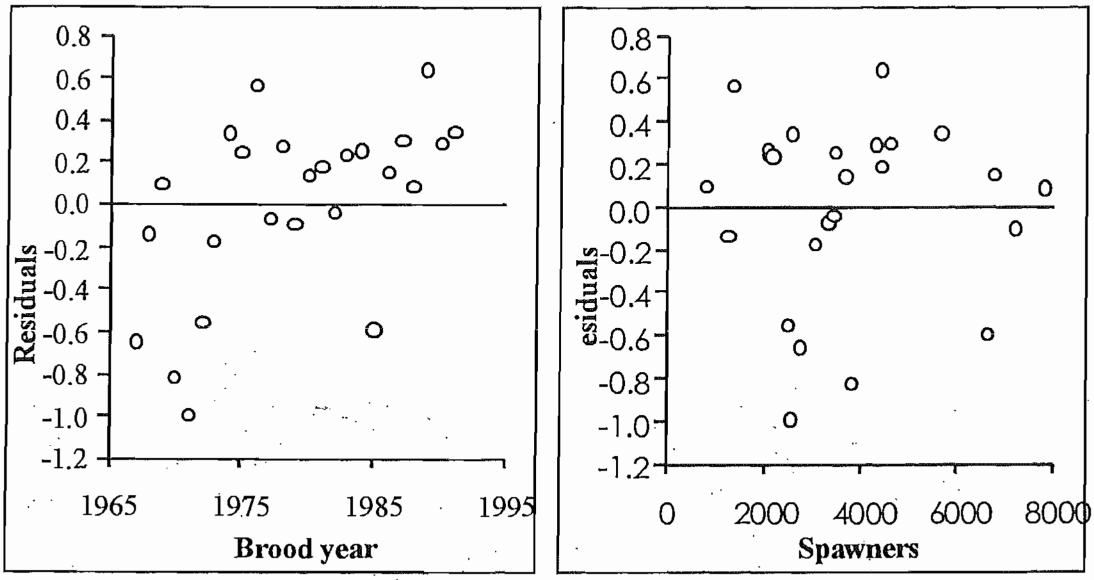

Figure 4.7. Residuals in the estimated spawner- recruit relationship for Siletz River fall chinook salmon, brood years 1967-1991. Residuals appear to change systematically over the time period. Estimated recruits may be biased low in the early years.

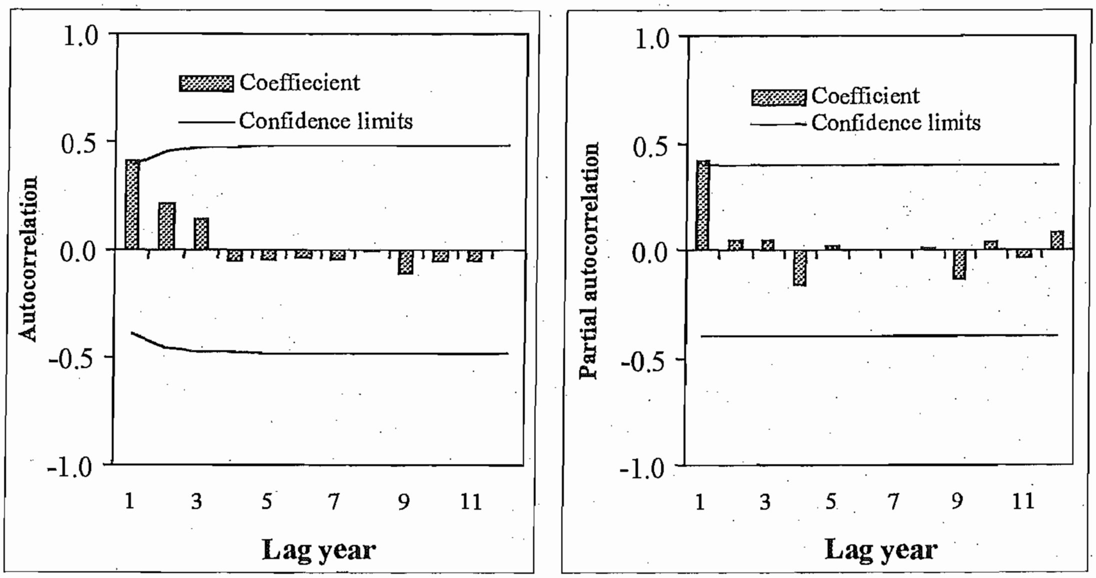

Figure 4.8. Autocorrelation and partial autocorrelation among residuals from the fit of Ricker's model to estimates of spawners and recruitment for brood years 1967-1991 for the Siletz River chinook salmon.

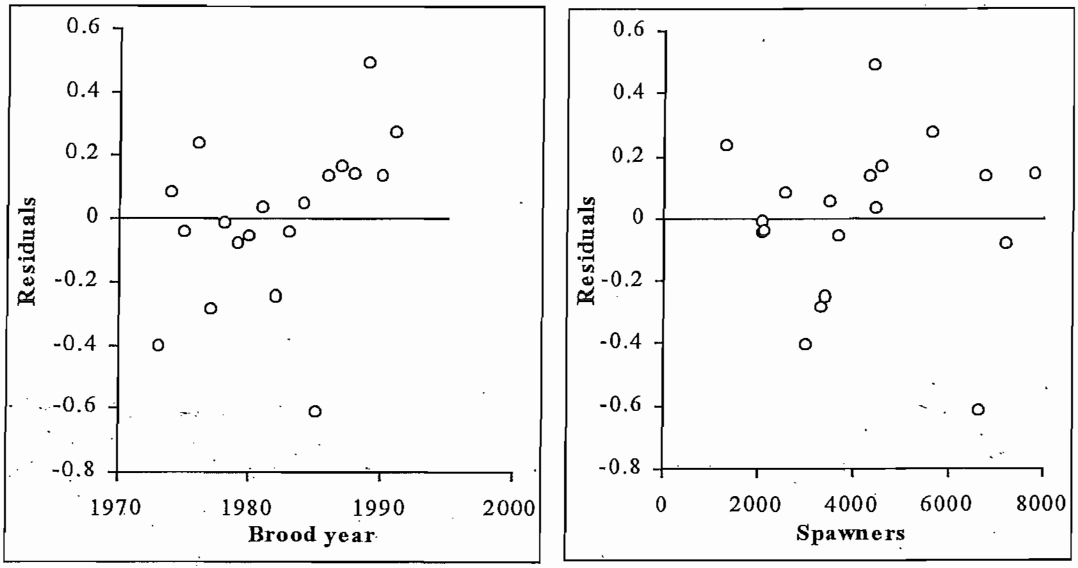

Figure 4.9. Residuals in the estimated spawner- recruit relationship for the Siletz River fall chinook salmon, brood years 1973-1991.

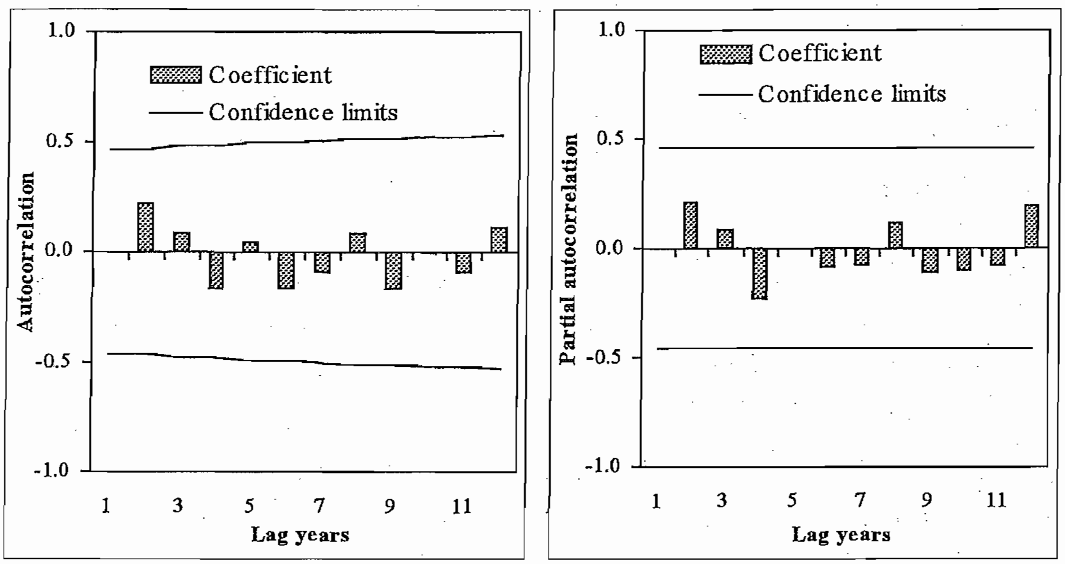

Figure 4.10. Autocorrelation and partial autocorrelation among residuals from the fit of Ricker's model to estimates of spawners and recruitment for brood years 1973-1991 for the Siletz River chinook salmon.

Table 4.10. Parameter estimates from the fit of Ricker's model to estimates of spawners and recruitment for the Siletz River fall chinook salmon, brood years 1973-1991.

	$\ln\hat{\alpha}$	$\hat{\beta}$	$\hat{S}_{MSY}$	$\hat{U}_{MSY}$	$\hat{S}_{MSP}$	$\hat{U}_{MSP}$
Model	2.493	0.000273	2,944	0.804	3,660	0.762
Bootstrap Mean	2.495	0.000273	2,975	0.804	3,716	0.760
SE	0.149	0.000034	294	0.025	475	0.036
CV	6%	12%	10%	3%	13%	5%
Lower 90% CI	2.244	3.32E-04	2,527	0.760	3,014	0.695
Upper 90% CI	2.743	2.21E-04	3,481	0.842	4,534	0.815

$$\ln\hat{\alpha} = 2.49;\ \hat{\beta} = 0.000273$$
$$R^2 = 0.806,\ \text{adjusted } R^2 = 0.795,\ n = 19,\ P < 0.001.$$

The residuals did not show abnormal patterns in this shorter time period (Figures 4.5-4.9 and 4.10).

Variance and Statistical Bias: The mean bootstrap estimate of S_{MSY} is 2,974 (Table 4.10), 31 fish more than the regression estimate of 2,944, indicating only a small statistical bias of 1.1%. The SE of this estimate from the bootstrap was 294 fish, representing a coefficient of variance of 10%. The mean estimate of S_{MSP} from the bootstrap was 3,716, which was 56 fish more than the regression estimate, indicating a statistical bias of 1.5% in the regression estimate; the coefficient of variation was 13%.

Management parameters: Estimated MSY occurred at about 3,000 spawners (90% CI = 2,527 to 3,481), and MSP was estimated at about 3,700 spawners (90% CI= 3,015 to 4,534) . The theoretical exploitation rates at MSY are and estimated 80% and at MSP an estimated 76%.

Stock Specific Discussion: Quality of data was very poor prior to 1986 and is elaborated on in the general discussion section of this chapter. In addition to the general uncertainties, time series biases resulted when analyzing this stock. We were able to obtain a better model fit and alleviate the time series bias by dropping the data from 1967-73 broods. The important question is – do we have evidence that these data may be biased or unsuitable for inclusion? There is no direct evidence that these data are not reliable, however we propose the following untested hypothesis as reason to exclude these data. There appears to be cyclic changes in the general ocean productivity that affect salmonid production in the Northeastern Pacific (Francis, R. C. and S.R. Hare, 1994; Mantua, N. J. et al, 1996 ; Pearcy, W. G., 1992 and Taylor, G. H., 1998). In reference to Oregon coho salmon, the shift in productivity seems to have occurred about 1976-1977. Prior to this period, coho salmon from the Pacific Northwest states benefited from good marine survival, while those from the Alaska gyre were less productive. Since the hypothesized regime shift in 1976-1977, these stocks have shown the opposite trends in the effect of marine survival. While the Pacific Northwest stocks have declined, the Alaska gyre populations have increased substantially.

We propose that the chinook populations may show similar trends linked to the major cycles in ocean productivity, but opposite of the coho populations. Chinook from the Oregon coast rear extensively in the Alaskan - Northern BC archipelago and would therefore be subject to good marine survivals when Alaskan coho stocks are also experiencing good marine survivals (a time when Pacific NW coho stocks show poor production). When the regime shifts, the Oregon Coastal chinook stocks are subject to poor marine survival as are Alaskan and North BC coho stocks. The period prior to 1976-1977 is hypothesized to have had good marine survival conditions for NW coho and therefore poor for NW chinook. After the regime shift NW coho had poor marine survival while Alaskan coho and NW chinook were subject to good survival. If we combined the stock- recruitment data from both survival regimes we are left with attempting to analyze a data set that is nonstationary, with the poor residual fit and autocorrelations observed. The non-stationarity can be exhibited when time is included as another variable in the general Ricker model. This exercise significantly improves the model fitting and the time variable is significant (P< 0.0001). Since it is difficult to interpret the effect of time we dropped this approach (Walters and Collie 1988, Hilborn and Walters 1992). Therefore we felt it justified to drop those years that could have had associated poor marine survival.

Consequently, we are left with an analysis that estimates the productivity of the stock under favorable marine survival conditions. If the regime shifts to a detrimental state for chinook marine survival, this analysis will be overly optimistic in estimating recruits from a given level of spawners, and overestimate the sustainable harvest rate. The management goals derived from this analysis are appropriate only if the recent favorable environmental regime continues. A revised management regime must be considered if the regime changes.

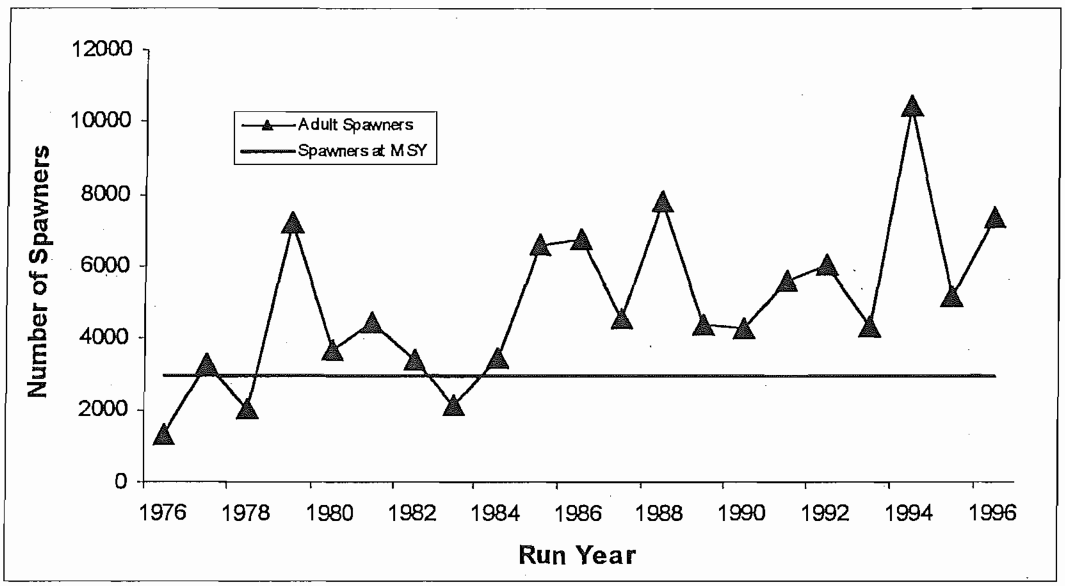

Figure 4.11. Escapement trend of Siletz River fall chinook in regard to estimated adult escapement for MSY. The goal is derived from data for brood years 1973-1991.

Since 1976, spawning escapements of adult (age 3-6) chinook have generally exceeded the MSY point estimate (Figure 4.11). In only three years did the escapement fall below the MSY point estimate, and these were all prior to 1984.

Stock 3. Siuslaw River Fall Chinook Salmon

Stock and Watershed Description: The Siuslaw River is a large coastal river system draining the coast range mountains along the central Oregon coast (Figure 4.1). The river drains a forested watershed of 588 square miles and includes approximately 890 miles of main-stem and tributaries. The entire watershed lies within a temperate rainforest climate, predominated by Douglas fir forests. A medium sized estuary provides a productive transition area for juvenile anadromous salmonids. Most of the watershed has been severely impacted by timber harvest. Residential development has been minor along the banks of the upper reaches of the river. Large-scale residential and commercial development has taken place adjacent to the estuary at the Town of Florence. Small scale agriculture, primarily livestock and dairy farms, exists along some of the riparian areas.

Although the Siuslaw River apparently supported a modest run of spring or summer run chinook in the early 1900s, the run today consists almost entirely of fall run fish. Juvenile fish have a predominance of zero age or "ocean type" juvenile rearing pattern. Juveniles sampled in the river from June to October were all under-yearlings, and scale patterns on returning adults showed that 100% migrated to the ocean as under-yearlings. Four release groups of hatchery chinook smolts revealed that they have a far north distribution similar to the North Oregon Coast exploitation rate indicator stock. Tag recoveries of the indicator stock occur predominantly in the SEAK and Northern BC fisheries. Very few recoveries are made in WCVI however a few more recoveries occurred in Oregon ocean fisheries than occurred with the exploitation rate indicator stock. A substantial number of fish are also caught in the terminal sport fishery in the estuary and river. Historically, this stock has not had an escapement goal, rather they were generally monitored to conform to a coastwide comprehensive escapement target density of 60-90 peak fish per mile as measured at standard survey sites.

Hatchery releases had little influence in the Siuslaw River. Historically five releases took place in this river. Twenty-two thousand to twenty-five thousand smolt were released from brood years 1978 to 1981. These fish were reared in private hatcheries and released in the Siuslaw River to mitigate for eggs removed from the natural spawning population (Nicholas and Hankin 1988). One more release of 10,494 fry occurred for brood year 1984 (J. Leppink, ODFW, Portland, personal communication). Because smolt and fry were released at least 10 river miles downstream of the standard survey sites, the influence of the hatchery fish is assumed insignificant and was not taken into account in the analysis.

Spawning Surveys: Spawning surveys have been conducted for chinook salmon on the river since 1952. There are eight standard survey sites encompassing 5.8 linear miles of prime chinook spawning habitat in the Siuslaw basin. Of these, five sites were not surveyed until 1987. One site, Lake Creek, supports an extremely high spawning population. When estimating the escapement, we separated this site (Lake Creek) from other sites. A recent study indicates that the total spawning habitat M is 237.9 miles in the Siuslaw River (S. Jacobs, ODFW Corvallis Research

Lab, personal communication). Of this total, four miles are in Lake Creek and the remaining 233.9 miles are distributed in the main-stem and other tributaries. An estimated detectability $\hat{p}$ of 0.5 is applied to the standard survey estimates. The data set was standardized to simulate the spawner densities across all 237.9 miles based on the random coho surveys in the river during 1990-1997. We derived an estimated correction factor $\hat{\phi} = 0.78$ (SD = 0.08) to apply to the standard chinook survey densities values. The spawning populations in year y are estimated for the Lake Creek and the other tributaries as separate groups as follows:

$$\text{For Lake Creek:} \quad \hat{S}_{y,L} = \hat{\bar{d}}_{y,L} M_L \hat{p}^{-1} \hat{\phi}$$

$$\text{For other tributaries:} \quad \hat{S}_{y,T} = \hat{\bar{d}}_{y,T} M_T \hat{p}^{-1} \hat{\phi}$$

where: $S_{y,L}$ = number of spawners in Lake Creek;
$S_{y,T}$ = number of spawners in other tributaries;
$\bar{d}_{y,L}$ = mean spawner density (fish/mile) in Lake Creek;
$\bar{d}_{y,T}$ = mean spawner density in other tributaries;
M_L = 4 miles, spawning habitat in Lake Creek;
M_T = 233.9 miles, spawning habitat in other tributaries.

The total spawners were the sum of spawners in Lake Creek and spawners in other tributaries. Lake Creek comprises some of the best spawning habitat in the basin.

Age Composition of the Adult Fish Escapement: Age data from scale analysis for Siuslaw adult fall chinook spawners have been available since 1986 (Borgerson and Bowden 1996). Estimates of age composition, determined from scales collected on the spawning grounds, were adjusted for size bias by using the estimated age compositions of the indicator stock at Salmon River. Age-4 and age-5 fish dominate the escapement with an estimated average 43% of adults age 4 (Table 4.11). Age-7 fish were very rare and were combined with age-6 fish for this analysis. The estimated average relative age composition was used to apportion the pre-1986 total escapement estimates into age classes.

Total Estimated Escapement: During 1965-1996, estimated density of adults density ranged from 19 to 876 fish/mile (Table 4.12) with a mean of 288 fish/mile during the 32 years (SD = 240). As a comparison, the estimated spawner density in the main-stem and other tributaries was only 2 - 103 fish/mile during the same period (mean = 32, SD = 25). On average, an estimated 12.9% of adult spawners (SD = 4.4%, n = 32) were found in Lake Creek. The estimated total escapement including both adults and jacks varied between 1,355 and 46,331 fish (mean = 15,694, SD = 10,730) in the entire river basin. On average, age-2 jacks composed an estimated 19.6% of total escapement (SD = 10.3%).

Table 4.11. Estimated relative age compositions of Siuslaw River fall chinook salmon. Data from carcasses recovered on spawning grounds were corrected with information from mark-recapture studies on the Salmon River.

Return Year	Age 3	Age 4	Age 5	Age 6+	Sample Size
1980	0.283	0.432	0.285	0.000	95
1981	0.265	0.554	0.182	0.000	43
1982	0.200	0.438	0.334	0.027	100
1983	0.284	0.477	0.213	0.026	36
1984	0.190	0.445	0.314	0.051	56
1985	0.051	0.520	0.384	0.045	185
1986	0.257	0.195	0.518	0.030	148
1987	0.305	0.541	0.114	0.040	204
1988	0.101	0.468	0.416	0.015	466
1989	0.300	0.289	0.407	0.005	520
1990	0.132	0.468	0.332	0.068	754
1991	0.154	0.403	0.411	0.032	331
1992	0.098	0.483	0.384	0.034	381
1993	0.179	0.191	0.599	0.031	313
1994	0.032	0.735	0.209	0.024	511
1995	0.290	0.171	0.520	0.018	275
1996	0.151	0.654	0.178	0.017	231
1997	0.083	0.296	0.618	0.002	483
Mean	0.186	0.431	0.357	0.026	285
SD	0.088	0.150	0.142	0.018	196

Freshwater Harvest: Like other coastal rivers, estimated annual freshwater harvests of chinook salmon have increased over time. A few hundred to slightly over a thousand fish were caught annually before the early 1980s (Table 4.13). Since 1988, estimated harvests increased to more than 2,000 per year and peaked in 1996 when more than 7,000 fish were caught. During 1965-1996, average estimated harvest was 1,779 (SD = 1,836), resulting in an estimated mean harvest rate of 9.2% (SD = 5.4%).

Recruitment: Recruitment, defined as total returns to the freshwater assuming no ocean fishery impacts, generally increased over the time period. Before brood year 1980, estimated recruits per brood were less than 30,000. From the 1980 brood to the 1991 brood, estimated recruitment increased to over or near 30,000 per brood except for brood year 1982 which had estimated recruits of 17,250 (Table 4.14). For the 27 broods from 1965-1991, the estimated ratio of total returns to spawners ($\hat{R}_y/\hat{S}_y$) varied between 0.8 and 21.1 (mean = 4.0, SD = 4.2).

Table 4.12. Estimated escapement by age of chinook salmon in the Siuslaw River from 1965 to 1996.

Estimates of relative age compositions from carcasses recovered on the spawning grounds were adjusted with information from mark-recapture studies on the Salmon River. The estimated mean relative age compositions between 1980-1997 were applied to run years 1965-1979. Lake Creek was separated from other sites because estimated fish densities (fish/mile) were consistently much higher than at other sites.

Estimated Spaner Densities	Lake Creek		Other Tribs		Estimated Escapement by Age					Estimated Escapement	
Run Year	Adult	Jack	Adult	Jack	Age 2	Age 3	Age 4	Age 5	Age 6	Adults	Total
1965	35.0	13.8	9.9	2.5	990	662	1,556	1,497	101	3,816	4,806
1966	138.8	13.8	23.7	4.6	1,759	1,648	3,873	3,725	251	9,497	11,257
1967	137.5	38.8	12.4	4.4	1,838	936	2,200	2,116	142	5,395	7,233
1968	65.0	40.0	7.5	5.0	2,068	547	1,285	1,236	83	3,151	5,219
1969	175.0	65.0	17.1	6.9	2,916	1,273	2,992	2,878	194	7,337	10,253
1970	320.0	95.0	30.2	11.7	4,875	2,261	5,313	5,110	344	13,028	17,904
1971	61.3	12.5	7.6	1.9	760	545	1,281	1,232	83	3,141	3,902
1972	110.0	70.0	7.8	4.8	2,201	614	1,443	1,388	93	3,539	5,740
1973	18.8	6.3	2.1	1.2	476	152	358	345	23	879	1,355
1974	163.8	85.0	13.7	6.9	3,037	1,044	2,453	2,359	159	6,015	9,052
1975	132.5	75.0	9.9	5.7	2,544	768	1,806	1,737	117	4,427	6,971
1976	235.0	92.5	17.9	8.1	3,522	1,388	3,262	3,137	211	7,999	11,520
1977	226.3	75.0	22.1	6.7	2,895	1,647	3,871	3,723	251	9,492	12,386
1978	143.8	30.0	13.6	3.0	1,282	1,019	2,395	2,303	155	5,872	7,153
1979	160.0	15.0	19.3	2.0	836	1,395	3,279	3,153	212	8,040	8,875
1980	272.5	30.0	24.5	5.6	2,248	3,006	4,597	3,027	0	10,630	12,878
1981	175.0	53.8	20.9	5.4	2,289	2,308	4,831	1,585	0	8,724	11,013
1982	257.5	42.5	25.4	4.9	2,047	2,179	4,762	3,631	299	10,870	12,917
1983	35.0	0.0	10.9	2.9	1,057	1,190	1,995	892	109	4,186	5,243
1984	128.8	8.8	28.4	4.8	1,801	2,126	4,966	3,510	565	11,168	12,969
1985	335.0	87.5	34.9	10.5	4,361	759	7,700	5,697	666	14,822	19,184
1986	318.8	85.0	35.2	13.5	5,440	3,813	2,895	7,686	450	14,844	20,283
1987	258.8	31.3	43.8	7.8	3,048	5,371	9,529	2,000	703	17,603	20,651
1988	672.5	65.0	102.9	11.5	4,585	4,222	19,521	17,383	621	41,746	46,331
1989	693.8	42.5	65.6	4.9	2,056	8,472	8,162	11,499	145	28,279	30,335
1990	722.5	53.8	61.1	6.2	2,591	3,530	12,555	8,901	1,814	26,799	29,391
1991	876.3	33.8	56.5	6.2	2,466	4,030	10,511	10,722	839	26,100	28,567
1992	651.3	40.0	60.4	4.5	1,908	2,564	12,599	10,029	898	26,090	27,998
1993	132.5	8.8	26.4	4.0	1,514	1,870	1,994	6,262	322	10,446	11,961
1994	375.0	23.8	58.2	6.4	2,470	760	17,313	4,927	570	23,570	26,040
1995	432.5	6.3	65.8	4.4	1,631	7,755	4,570	13,904	486	26,715	28,346
1996	767.5	36.3	77.5	3.3	1,420	4,976	21,630	5,889	557	33,051	34,471

Table 4.13. Estimated freshwater harvest by age and freshwater harvest rates for Siuslaw River fall chinook, 1967-1996.

Relative age composition estimated from the Salmon River creel census is used to apportion harvests from the Siuslaw River. The estimated mean relative age composition between 1986-1996 is applied for run years 1967-1985.

Run Year	Age 2	Age 3	Age 4	Age 5	Age 6	Estimated Total Catch	Estimated Harvest Rate
1967	137	193	294	301	33	957	0.117
1968	67	94	142	146	16	464	0.082
1969	75	106	161	165	18	524	0.049
1970	75	106	161	165	18	524	0.028
1971	38	53	80	82	9	262	0.063
1972	110	154	234	239	26	763	0.117
1973	31	44	67	69	8	219	0.139
1974	23	33	50	51	6	162	0.018
1975	19	27	41	42	5	134	0.019
1976	24	34	52	54	6	171	0.015
1977	138	194	295	302	33	963	0.072
1978	160	225	342	350	38	1,115	0.135
1979	121	169	258	264	29	841	0.087
1980	88	123	188	192	21	612	0.045
1981	98	138	210	215	24	684	0.059
1982	144	202	307	314	34	1,001	0.072
1983	111	156	237	243	27	774	0.129
1984	202	283	431	441	48	1,405	0.098
1985	156	219	334	342	37	1,089	0.054
1986	312	248	282	444	55	1,340	0.062
1987	98	480	491	694	73	1,837	0.082
1988	198	48	1,710	645	108	2,711	0.055
1989	2,409	756	283	2,149	71	5,669	0.157
1990	438	1,161	948	355	107	3,010	0.093
1991	609	341	1,816	1,342	21	4,128	0.126
1992	418	308	571	1,040	205	2,542	0.083
1993	130	1,459	1,698	869	130	4,286	0.264
1994	473	110	1,452	795	41	2,870	0.099
1995	518	1,211	413	2,864	134	5,141	0.154
1996	305	3,223	2,453	795	406	7,182	0.172

Table 4.14. Estimated spawners and estimated recruits for the Siuslaw River fall chinook stock, brood years 1965-1991.

Brood Year	Spawners $\hat{S}_y$	Recruitment: Age 2	Age 3	Age 4	Age 5	Age 6	$\hat{R}_y$	$\hat{R}_y/\hat{S}_y$	$\ln(\hat{R}_y/\hat{S}_y)$
1965	3,816	2,039	744	4,513	12,154	308	19,757	5.2	1.644
1966	9,497	2,217	1,600	7,957	2,882	388	15,044	1.6	0.460
1967	5,395	3,085	2,732	1,928	3,536	93	11,374	2.1	0.746
1968	3,151	5,127	690	2,373	870	484	9,542	3.0	1.108
1969	7,337	822	880	589	5,088	355	7,735	1.1	0.053
1970	13,028	2,388	225	3,490	3,434	579	10,115	0.8	-0.253
1971	3,141	523	1,227	2,495	6,408	799	11,452	3.6	1.294
1972	3,539	3,153	898	4,504	9,256	766	18,576	5.2	1.658
1973	879	2,630	1,617	5,964	7,242	1,098	18,551	21.1	3.050
1974	6,015	3,653	2,121	4,125	9,224	120	19,243	3.2	1.163
1975	4,427	3,132	1,455	5,355	8,905	118	18,966	4.3	1.455
1976	7,999	1,494	1,830	7,558	5,044	1,548	17,474	2.2	0.781
1977	9,492	1,001	3,947	9,551	10,417	639	25,555	2.7	0.990
1978	5,872	2,449	3,310	10,516	5,103	4,923	26,302	4.5	1.500
1979	8,040	2,482	2,771	3,260	10,258	3,260	22,030	2.7	1.008
1980	10,630	2,269	1,593	7,828	16,147	2,412	30,250	2.8	1.046
1981	8,724	1,204	2,860	11,946	21,026	4,284	41,320	4.7	1.555
1982	10,870	2,054	1,165	4,845	6,766	2,420	17,249	1.6	0.462
1983	4,186	4,600	4,379	12,354	31,000	1,266	53,599	12.8	2.550
1984	11,168	5,929	6,492	27,801	29,880	5,826	75,928	6.8	1.917
1985	14,822	3,280	4,746	10,600	19,226	4,114	41,966	2.8	1.041
1986	14,844	4,940	10,208	17,447	29,734	6,595	68,924	4.6	1.535
1987	17,603	4,633	5,388	17,140	22,224	2,552	51,937	3.0	1.082
1988	41,746	3,155	4,994	19,627	21,508	2,604	51,888	1.2	0.217
1989	28,279	3,217	3,258	5,349	13,705	2,028	27,557	1.0	-0.026
1990	26,799	2,381	3,755	26,006	29,577	3,993	65,711	2.5	0.897
1991	26,100	1,701	1,003	7,207	15,875	863	26,649	1.0	0.021

Stock-Recruit Analysis: We examined both Ricker and Beverton-Holt models to fit the spawner-recruit data (Hilborn and Walters 1992). Because the Ricker model resulted in a better fit, we adopted this model (Figure 4.12).

$$\ln \hat{\alpha} = 1.577;\ \hat{\beta} = 0.0000443$$
$$R^2 = 0.304,\ \text{adjusted } R^2 = 0.276,\ n = 27,\ P = 0.0029.$$

Residuals were examined and did not show abnormal patterns (Figure 4.13).

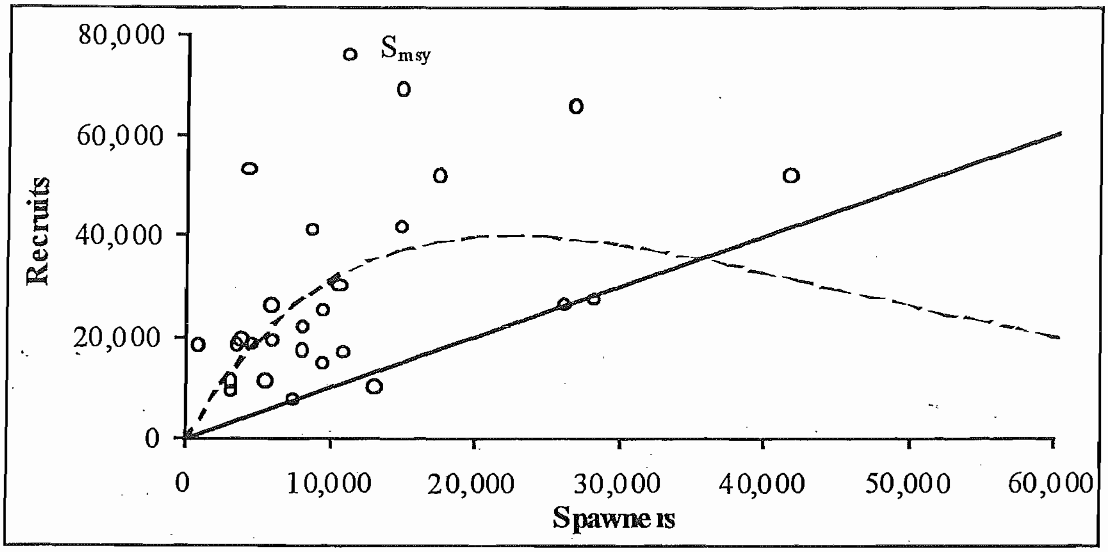

Figure 4.12. Estimated spawner-recruit relationship for Siuslaw River fall chinook during brood years 1965-1991.

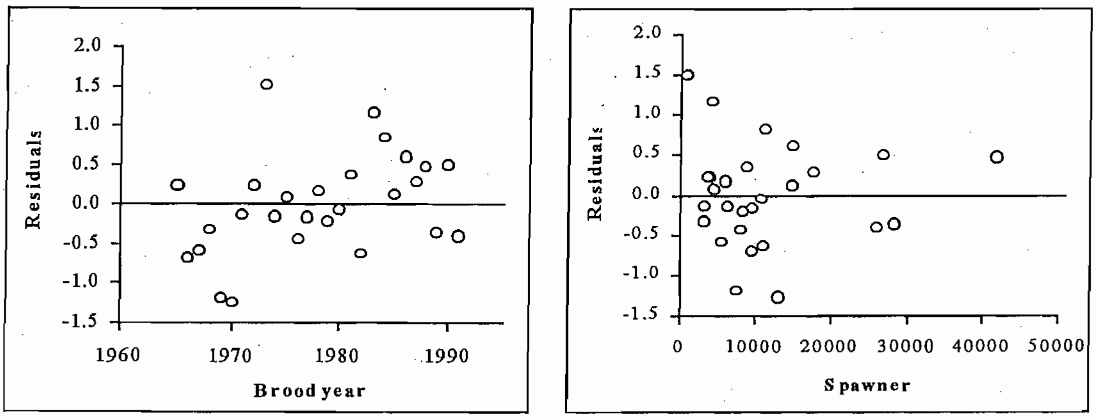

Figure 4.13. Residuals in the estimated spawner-recruit relationship for the Siuslaw River fall chinook salmon stock, brood years 1965-1991.

No significant autocorrelation and partial autocorrelation were detected (Figure 4.14).

<u>Variance and Statistical Bias:</u> The mean bootstrap estimate of S_{MSY} was 13,991 and is 1,066 fish more than the regression estimate, indicating a statistical bias of 8.3% (Table 4.15). The SE of $\hat{S}_{MSY}$ from the bootstrap was 6,824 fish, representing a coefficient of variation of 49%. The mean bootstrap estimate of S_{MSP} was 25,199, which was 2,648 fish more than the regression estimate, indicating a statistical bias of 11.7% in the regression estimate. The coefficient of variation for $\hat{S}_{MSP}$ was 60 %.

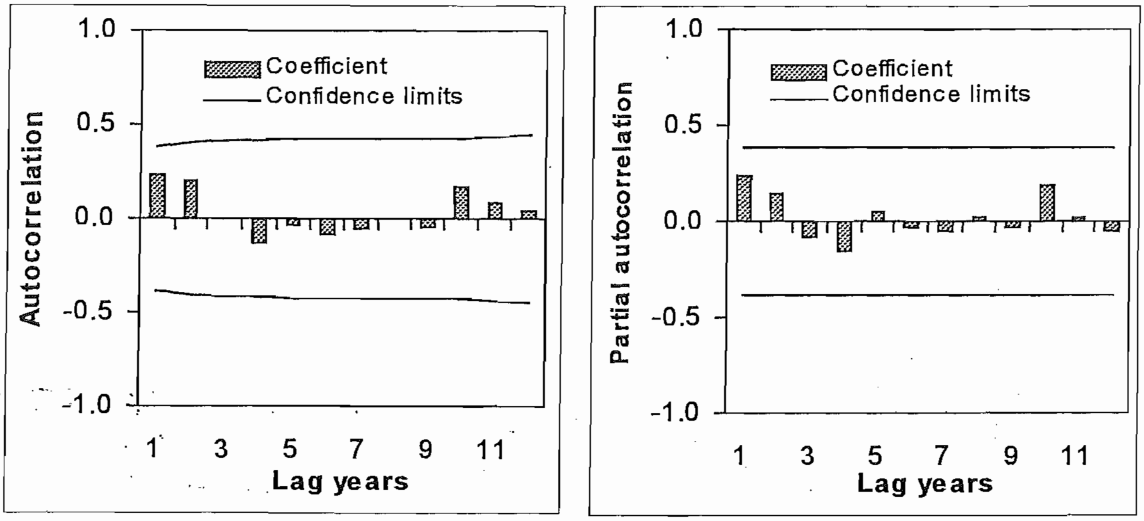

Figure 4.14. Autocorrelation and partial autocorrelation among residuals from the fit of Ricker's model to estimates of spawners and recruitment for brood years 1965-1991 for Siuslaw River fall chinook salmon.

Table 4.15. Parameter estimates from the fit of Ricker's model to estimates of spawners and recruitment for the Siuslaw River fall chinook salmon, brood years 1965-1991.

		$\ln\hat{\alpha}$	$\hat{\beta}$	$\hat{S}_{MSY}$	$\hat{U}_{MSY}$	$\hat{S}_{MSP}$	$\hat{U}_{MSP}$
Model		1.577	0.000044	12,925	0.573	22,551	0.346
Bootstrap	Mean	1.587	0.000045	13,991	0.612	25,199	0.400
	SE	0.196	0.000013	6,824	0.053	15,032	0.119
	CV	12%	29%	49%	9%	60%	30%
	Lower 90% CI	1.259	6.64E-05	9,541	0.519	15,054	0.184
	Upper 90% CI	1.908	2.40E-05	20,958	0.695	41,630	0.573

Management Parameters: According to this analysis, the estimated escapement goal for MSY should be about 13,000 adult spawners, (90% CI = 9,541 to 20,958) and MSP occurred at about 23,000 spawners (90% CI = 15,054 to 41,630). Since the Lake Creek spawners comprised an estimated 12.9% (SD = 4.4% between 1965-1996) of the total adult spawners on average, the MSY escapement goal should be about 1,680 adults for Lake Creek and 11,320 adults for other tributaries.

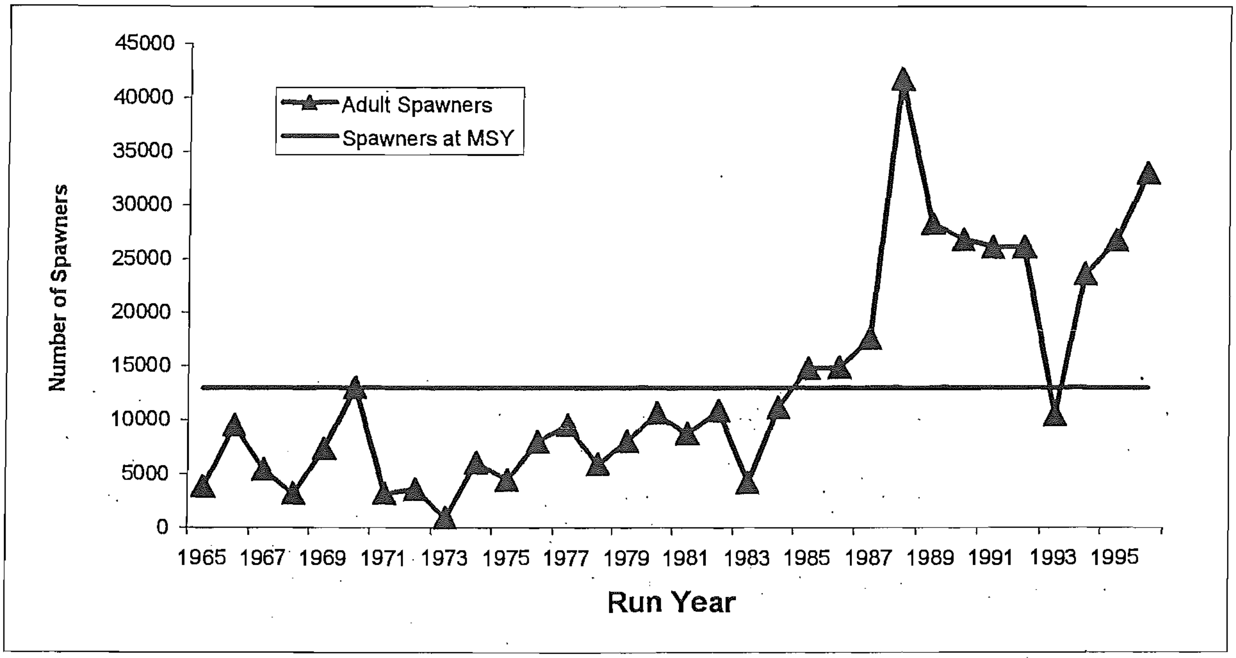

Figure 4.15. Escapement trends for Siuslaw River fall chinook salmon relative to estimated escapement that would produce MSY.

Stock Specific Discussion: Given the interim escapement goal of 13,000 adult spawners to achieve MSY, the escapements over the last 32 years have shown a very unbalanced trend around this goal (Figure 4.15). For 19 of 20 years during the period 1965-84, escapements were always below the MSY point estimate. Since 1985 escapements have always been greater than the MSY point estimate except in 1994. It appears either there has been a major shift in the environmental regimes affecting this stock or serious flaws in the monitoring program resulting in discontinuous data.

Because the shift from low escapements to higher levels in the Siuslaw River occurred in 1985, it does not follow the hypothesized regime shift that we proposed earlier for the Siletz stock is the cause of the shift in the Siuslaw . Prior to 1985 very few sites were surveyed while after that year additional sites were incorporated in the annual spawning surveys for chinook. Between 1975 and 1987 only three standard sites encompassing 2.6 miles were surveyed. This represented only 1.1% of the potential identified spawning area. Since 1987 eight sites have been surveyed, totaling 6.3 miles, or 2.7% of the potential spawning area. This may be far too few surveys to yield a reliable estimate of spawner density to be used for this analysis, resulting in poor estimates of spawners and subsequently recruits as well.

Nonetheless, those surveys that were made during both time periods show a definite increase in chinook counts after 1987, particularly for Lake Creek. There was approximately an order of magnitude difference in spawner densities in Lake Creek between the pre- and post -1987 surveys. This general trend appears also in the other surveys sites but the scale of difference is not so large (see Table 4.12). No explanation has been provided for this change, and the change does not appear in any of the other coastal river basins to this magnitude.

Annual escapements may have been overestimated. One way to check this assumption comes from estimates freshwater harvest. Within an individual Oregon coastal river basin, freshwater harvest is generally believed between 20 and 30% of abundance. This is true for the Nehalem, Siletz, Elk, and Salmon Rivers. However, from 1965-1996, the mean freshwater harvest rate for the Siuslaw is only 9.2%. Although the estimated harvest rate increased to over 15% in recent years, this rate is still lower than that of the other coastal rivers. This may simply be the result of lower fishing pressure in the Siuslaw River system, or, it may be the result of either underestimating the freshwater catch or overestimating the terminal run size. Because freshwater catch is estimated from the same punch card database as used for the Nehalem and Siletz Rivers, overestimating the terminal run size seems more likely.

For these reasons we have a chinook stock that shows problems when we use the historical database, and the analysis is therefore suspect. Unfortunately, because of the unreliability of the historic monitoring program, the cause of this phenomenon is not known, whether it is environmental or unreliable data.

Because the analysis has numerous identified data problems, we suggest that this stock be included in the CTC escapement goal chapter as a stock with a preliminary MSY escapement goal. Although we cannot re-monitor the past runs, improved monitoring data currently being collected will assist in the future in redefining an appropriate escapement goal for this stock.

4.3. General Discussion of Oregon Coastal Fall Chinook Analyses

The accuracy and precision of escapement data has the most significant influence on stock recruitment analysis because not only are the spawners but also the recruits derived from escapement. There are several potential problems associated with these escapement data as used in our abundance estimation procedure. Chinook index spawning surveys have been conducted since the early 1950s in coastal Oregon rivers. Surveys are conducted and the peak fish count is used as the annual measure of abundance at the index sites. Factors that influence the peak count other than fish abundance include: observer experience, water clarity, predator removal of dead fish, and site location as live fish in lower survey sites can be doubly counted in upstream survey sites at a later date. We used a detectability factor of 0.5 to convert the peak count to the total assumed number of chinook that would use a survey site during the entire season. These measures need to be investigated to verify their use in estimating the abundance of spawning chinook. To improve the quality of escapement information, we strongly recommend the following research in the future:

(1) Calibrate the historical standard survey data. While continuing standard index and random surveys, estimate the total escapement using other methods (e.g., mark-recapture, fish weir count, sonar census, etc.) to derive the appropriate measurement unit and expansion factors (detectability and spawning mileage) for the coastal rivers.
(2) Re-design the survey scheme. To increase the accuracy and obtain variance estimates, a statistically designed sampling plan with a reasonable sample size should be

employed in the future. Random or stratified random surveys will be more effective in estimating spawner abundance than the selected index surveys.

If ignored during a stock-recruit analysis, measurement error in estimates of spawning escapements and recruitment will produce biased estimates of S_{MSY} and other management parameters. Measurement error in estimates of spawning abundance biases $\hat{S}_{MSY}$ downward and $\hat{U}_{MSY}$ upward, while measurement error in estimates of recruitment interferes with the usual adjustment in $\hat{S}_{MSY}$ for process error. Methods are available to adjust estimates of management parameters for measurement error if estimates of variance are available for statistics on spawning abundance and recruitment. Unfortunately, no such estimated variances are available for stocks in the Nehalem, Siletz, and Siuslaw rivers, so estimates of management parameters remain unadjusted for measurement error. Fortunately, ODFW has implemented new, statistically rigorous studies with funds from the U.S. Sections of the Pacific Salmon Commission to provide accurate estimates of spawning abundance with measurable precision. These funded studies should provide the needed information over the next several years to estimate management parameters with a degree of acceptable reliability.

The punch card reporting program was designed in the early 1960s (Hicks and Calvin 1964). Evolution of the fisheries, harvest regulations, and reporting incentives, may have influenced the suitability of the non-report bias correction factor currently used. This in turn would effect the comparison between years of the freshwater harvest to accurately estimate the true catch. In addition, one uniform expansion factor has been applied to coast wide harvest. More accurate catch estimates such as estimates from creel surveys for individual river basins would be helpful. Again, some funding from the U.S. Section of the Pacific Salmon Commission is being used to research this area.

Using scales from fish carcasses recovered on the spawning grounds for age estimation underestimates small fish (ages 2 and 3) while overestimating large fish (ages 4, 5, and 6). Small-sized carcasses may be more likely to be consumed by scavengers, more difficult for samplers to detect, and more readily washed away. By using Salmon River mark-recapture information, we assume that the detectability for the same size class between Salmon River and other coastal rivers is the same. This assumption may be violated due to habitat and water condition differences. Furthermore, the Salmon River mark-recapture studies were designed for estimating total population rather than correcting age bias. Some funding from the U.S. Section of the Pacific Salmon Commission is currently being used to research selectivity of carcass collections.

Because actual age composition data from the spawning escapement is only available beginning in 1985, analysis with the most complete data set would allow only 9 brood years to be investigated. This was a very short series to use in a stock and recruitment analysis and we chose to expand the data set by including data back to the 1967 return year. Consequently, we were able to include from 19 to 27 brood years in these analyses. However, the lack of age composition information prior to 1985 required that we applied estimated average relative age composition to the annual spawning escapement for those years. This estimation procedure will mask the actual inter-annual variation in age specific escapements and result in inaccurate estimates of recruits. The extent to which this influences the results of the analysis is unknown, but caution must be taken when interpreting these results to describe the true productivity of the stock. Nonetheless, we felt it advantageous to include this data in the analysis at this time to provide a data series sufficiently large to conduct this analysis. Improvements are needed for estimating the age composition of naturally spawning stocks, for example, developing correction factors by age rather than size, increasing sample size, etc.

These data limitations and problems are prevalent throughout the Oregon coastal wild stocks of chinook salmon as similar spawning survey and catch reporting methods are being used on all major rivers. Under the newly agreed PSC abundance-based management scheme, the escapement monitoring program for Oregon coastal wild stocks needs improvement to meet the data requirement for high resolution of production and harvest. Data shortcomings as discussed above are being researched with U.S. Section Pacific Salmon Commission funding and these needs should continue to be taken into account when funding new coast-wide research to meet the demands of the treaty.

Despite these uncertainties, this analysis can provide managers with information to establish interim management goals and design annual harvest arrangements under the current treaty agreement. We feel that the results of this paper are satisfactory until more precise stock specific data are available. Management policies regarding specified escapement goals (point estimates or ranges) have not been determined in this analysis, and will be subject to further discussion and analysis by the CTC in consultation with agency managers.

There was high degree of uncertainty in estimating these parameters. A variety of environmental factors affect the survival of chinook salmon regardless of their abundance. A single level of spawning abundance does not always produce the same level of production, but rather, a range of production (CTC, 1998). Also, a high degree of measurement and process errors may exist in the stock-recruit analysis (Ludwig and Walters 1981, Hilborn and Walters 1992). Considering these uncertainties, a range for a management parameter should be more appropriate than a point estimate.

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APPENDIX A.

Appendix A.1. Indicator Stocks to Estimate Production

Chinook salmon produced by a given spawning escapement mature at different ages and are harvested at various stages of maturity. With few exceptions, direct estimates of pre-terminal fishery impacts are rarely available for naturally-produced "wild" stocks. Absent such data, information derived from CWT experiments for an exploitation rate "indicator" stock that is representative of a natural stock of interest are often used. Results of cohort analysis on CWT data for the exploitation rate indicator stock can be employed to estimate production of natural stocks for stock-recruit analysis. Cohort analyses techniques generate estimates of survival rates, age-specific fishery exploitation rates, and maturation schedules for individual brood years for indicator stocks. The general methodology is outlined below.

The simplest approach to estimate production would be to divide estimates of spawning escapements (adjusted for pre-spawning mortality) for a wild stock by (1- the total adult equivalent exploitation rate) for the appropriate indicator stock. The total adult equivalent exploitation rate for both stocks would be estimated from cohort analysis based on CWT recovery data for the indicator stock.

A more involved procedure could be employed if age-specific estimates of spawning escapements of acceptable accuracy are available for the wild stock. For each age:

1. Divide spawning escapement estimates for the wild stock by post-fishery, pre-spawning survival rates from the indicator stock to estimate survival past fisheries for "wild" fish.

2. Divide the result of step 1 by the complement of the terminal harvest rate (1-terminal harvest rate) estimated from cohort analyses of the appropriate brood for the indicator stock to estimate the mature run size for the wild stock.

3. Divide the result of step 2 by the maturation rate estimated for the indicator stock to estimate the number of "wild" fish surviving after pre-terminal fisheries.

4. Estimate the pre-terminal mortality (catch plus incidental fishing mortality) of "wild" fish by multiplying the result of step 3 by (pre-terminal exploitation rate)/(1-pre-terminal exploitation rate) with rates estimated for the indicator stock.

5. Convert pre-terminal mortality to adult equivalents for the wild stock by multiplying the result of step 4 by the adult equivalent factor for the indicator stock (see following section).

Total production from the "wild" brood is then the sum by age of the terminal run sizes (step 2) plus the adult equivalent catches (step 5).

In the example below, estimated rates correspond to the indicator stock and estimated abundance to the wild stock:

	Age 2	Age 3	Age 4	Age 5
Spawning Escapement For Brood Year	100	200	175	40
Post-fishery, Pre-spawning Survival Rate	0.90	0.90	0.95	0.95
STEP 1 (Post-fishery Escapement)	111	222	184	42
Terminal Harvest Rate	0.10	0.25	0.40	0.30
(1- Terminal Harvest Rate)	0.90	0.75	0.60	0.70
STEP 2 (Terminal Run Size)	123	296	307	60
Maturation Rate	0.02	0.30	0.75	1.00
STEP 3 (Ocean abundance after Pre-terminal Fisheries)	6173	988	409	60
Pre-terminal Total Exploitation Rate	0.05	0.20	0.35	0.40
STEP 4 (Pre-terminal Mortality)	325	247	220	40
Adult Equivalent Factor	.600	.8460	.975	1.00
STEP 5 (Adult Equivalent Pre-terminal Mortality)	195	209	215	40

The estimated production for the brood year is 1445, the sum of the values from Step 5 (adult equivalent pre-terminal mortality) plus the sum of the values from Step 2 (Terminal Run Size).

Note that the above procedure can produce a reconstructed harvest and escapement that may not be totally consistent with the results of a cohort analysis. For instance, the ocean abundance for fish age a after pre-terminal fisheries (the result of Step 3) discounted for natural mortalities should be similar to the sum of ocean abundance before (Step 3) and mortalities during (Step 4) pre-terminal fisheries a year later for fish age a+1. Differences in comparable statistics arise for several reasons, including: (a) error in estimates of spawning escapements by age; (b) variability in sampling for CWTs; (c) errors in assumed parameters for cohort analyses.

In mathematical terms, the five steps can be expressed as:

$$\hat{R}_y = \sum_a \hat{S}_{y(a)}\left[\hat{\pi}_1\hat{\pi}_2 + \hat{\pi}_1\hat{\pi}_2\hat{\pi}_3\hat{\pi}_4\hat{\pi}_5\right] = \sum_a \hat{S}_{y(a)}\hat{\Pi}_a ,$$

where $\hat{R}_y$ is the estimated production from brood year y of "wild" fish, $\hat{S}_{y(a)}$ is the estimated abundance of "wild" fish in brood year y spawning at age a, and the $\{\hat{\pi}_i\}$ correspond to expansion factors for each of the five steps above and are based on harvest and maturation rates estimated for the appropriate indicator stock and the adult equivalent factor. The estimated variance for estimated production is:

$$v(\hat{R}_y) = \sum_a [\hat{S}^2_{y(a)} v(\hat{\Pi}_a) + (\hat{\Pi}_a)^2 v(\hat{S}_{y(a)}) - v(\hat{\Pi}_a) v(\hat{S}_{y(a)})] .$$

Because $\hat{S}_{y(a)}$ and $\hat{\Pi}_a$ are obtained from independent sampling programs (different programs on different stocks in the same or different years), no covariances are involved in the estimation of variance above. However, covariances can be involved in estimating the variance for $\hat{\Pi}_a$:

$$v(\hat{\Pi}_a) \cong \sum_i v(z_i)\left[\frac{\partial\hat{\Pi}_a}{\partial z_i}\right]^2 + 2\sum_{i>j}\sum \mathrm{cov}(z_i, z_j)\left[\frac{\partial\hat{\Pi}_a}{\partial z_i}\right]\left[\frac{\partial\hat{\Pi}_a}{\partial z_j}\right],$$

where the $\{z_i\}$ are the harvest and maturation rates and adult equivalent factors estimated for the indicator stock. The covariances arise because some of the $\{z_i\}$ are estimated from the same cohort analysis. The relationship between expansion factors and estimated rates along with the partial derivatives used in the approximate variance above are given in the following table:

Expansion	Definition	$\frac{\partial\hat{\Pi}_a}{\partial z_i}$
$\hat{\pi}_1$	Reciprocal of the post-fishery, pre-spawning survival rate $[z_1^{-1}]$	$-\hat{\pi}_1\hat{\Pi}_a$
$\hat{\pi}_2$	Reciprocal of the complement of the terminal harvest rate $[(1-z_2)^{-1}]$	$\hat{\pi}_2\hat{\Pi}_a$
$\hat{\pi}_3$	Reciprocal of the maturation rate $[z_3^{-1}]$	$\hat{\pi}_1\hat{\pi}_2\hat{\pi}_3^2\hat{\pi}_4\hat{\pi}_5$
$\hat{\pi}_4$	The pre-terminal exploitation rate divided by its complement $[z_3(1-z_3)^{-1}]$	$\hat{\pi}_1\hat{\pi}_2\hat{\pi}_3\hat{\pi}_5$
$\hat{\pi}_5$	The adult equivalent factor $[z_5]$	$\hat{\pi}_1\hat{\pi}_2\hat{\pi}_3\hat{\pi}_4$

The next table corresponds to values for $\{z_i\}$ and $\{\hat{\pi}_i\}$ for age-2 chinook salmon corresponding to the five steps in the example above. The table also contains values of the first partial derivatives for this age group, given that $\hat{\Pi}_{a=2} = 3.184 = 1.111(1.111)50(0.053)0.60$.

i	z_i Definition	Value	$\hat{\pi}_i$	$\frac{\partial\hat{\Pi}_a}{\partial z_i}$
1	Post-fishery, pre-spawning survival rate	0.90	1.111	-3.538
2	Terminal fishery harvest rate	0.10	1.111	3.538
3	Maturation rate	0.02	50.000	97.466
4	Pre-terminal fishery exploitation rate	0.05	0.053	37.037
5	Adult equivalent factor	0.60	0.600	3.249

Similar calculations are required for ages-3 through 5 to provide statistics for estimating variance of $\hat{\Pi}_a$. Estimated variances and covariances for and among the $\{z_i\}$ can be obtained through a cohort analysis based on the return of CWTs from the indicator stock.

Appendix A.2. Adult Equivalent Mortalities

Estimating production for stocks whose members are subject to exploitation as immature fish involves adjusting harvests to represent mature fish. "Adult equivalent mortalities" are calculated as a combination of maturity and survival rates applied to estimated harvests. From PSC (1988: Appendix B, p.7)

$$\hat{H}'_{y(a)} = \hat{H}_{y(a)} \dot{e}_{y(a)}$$

$$\dot{e}_{y(a)} = \dot{m}_{y(a)} + (1 - \dot{m}_{y(a)}) \dot{s}_{a+1} \dot{e}_{y(a+1)}$$

where $\hat{H}_{y(a)}$ is the unadjusted estimate of harvest and $\hat{H}'_{y(a)}$ the adjusted for year class y at age a, $\dot{e}_{y(a)}$ is the estimated adjustment rate for this group, $\dot{m}_{y(a)}$ is the estimated maturity rate, and $\dot{s}_{a+1}$ is the survival rate for year class y from age a to a+1. The hierarchical solutions to the above equations are constrained by $\dot{e}_{y(\max)} = 1$ (all harvests in the oldest age group represent mature fish). The estimate of variance for the adjusted harvest estimate (the adult equivalent mortalities) is not straight forward, but can be approximated through simulation.