CHARACTERIZING AND ACCOUNTING FOR UNCERTAINTIES IN PACIFIC SALMON FISHERIES

by

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Abstract

Uncertainties are widespread in Pacific salmon fisheries, occurring in both biological and management components. Despite recent attempts by scientists to account for uncertainties in stock assessments and incorporate them into advice to managers, some sources of uncertainties are often still ignored. My dissertation addressed four of these sources for Pacific salmon fisheries related to: forecasts of adult abundances, effectiveness of fishing regulations when implemented, the ability of management to achieve stated objectives, and international management institutions.

First, despite evidence for long-term increases in age-at-maturity of sockeye salmon (*Oncorhynchus nerka*), these increases have not been accounted for when forecasting age-specific abundances of adult returns. To account for these trends, I developed a new forecasting model that used a Kalman Filter to track long-term changes in its parameters.

Second, stock assessors and managers often assume that management targets are achieved exactly, but outcomes from fishing regulations rarely match targets. I quantified bias and imprecision in this "outcome uncertainty" for sockeye salmon fisheries on the Fraser River, British Columbia, and developed a method to adjust current harvest rules for that source of uncertainty.

Third, although my research suggests that these two modifications to forecasting models and harvest rules are feasible, the magnitude of improved performance of
management that includes these modifications is not certain. I evaluated the performance of these two modifications in a computer model for sockeye salmon fisheries in British Columbia and Alaska. I found that, compared to the status quo, performance increased when outcome uncertainties were accounted for, but in contrast to initial expectations, this was not the case when accounting for forecasting uncertainty.

Fourth, salmon body sizes at a given age have declined across the North Pacific over the last fifty years, in part due to increased competition among salmon populations from different countries for a common pool of prey resources. However, the incentives for collective action to reverse this trend are not clear, creating uncertainties in resolving this common-pool resource problem. I investigated the potential for international cooperation on this problem by identifying the incentives and disincentives for altering behaviour of individual nation states to encourage collective action.

**Keywords:** sockeye salmon; management; uncertainty; simulation modelling; international policy

**Subject Terms:** Fishery management; Fish stock assessment; Natural resources -- International cooperation
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General Introduction

Uncertainties are ubiquitous in fisheries and they create widely recognized biological risks for fish populations and economic and social risks for harvesters. Although fisheries management has made progress in addressing these risks through risk assessments and quantitative decision analysis (Peterman 2004), important challenges remain. A better understanding of the magnitude and nature of uncertainties and the scales at which they occur may help managers account for them when making decisions, potentially reducing those risks.

Categories of uncertainties

Uncertainties can be divided into at least three categories, those related to random fluctuations ("noise"), imprecise knowledge of parameter values or state variables (e.g., abundance estimates), and ignorance of the overall structure of the system (Walters 1986; Charles 1998). The first category, random fluctuations, includes variability in space and time, across individuals in the biological system, and in the management system (due to, for example, imprecise measurements). This category is relatively well understood by scientists and is often represented by probability distributions. Fisheries managers can account for random fluctuations when choosing harvest policies by using control systems that incorporate feedback between the choice of a policy and effects on management performance that vary with random fluctuations. Specifically, control systems are decision algorithms that seek to maintain an attribute of the system such as abundance of
recruits or catch at a desired level given random noise in system components (de la Mare 2005). Stochastic optimizations can be used to estimate the form of the decision algorithm that best achieves objectives (e.g., Clark 1990), and simulation models can be used to compare performance of alternative algorithms (e.g., Cooke 1999). However, uncertainties that are not taken into account in optimizations or simulation models may thwart managers' efforts to control attributes of the system and achieve management objectives. In particular, the combined effects of uncertainties from various components of the fisheries system are often not obvious, representing a gap in our knowledge about the overall effects of management actions on performance (Charles 1998). For example, natural variability in population abundances can propagate through biological and management systems causing deviations in outcomes of fishing regulations from targets, but these effects are often overlooked when choosing harvest policies and evaluating management options.

The second category of uncertainty, imprecise knowledge of parameter values and state variables, arises because of scientists' inability to get perfect information (due to, for example, sampling error), and provides even greater challenges than the previous category. For example, the parameters of the relation between spawners and adult abundances in the subsequent generation are often estimated with imprecision and bias because of natural variability in productivity and measurement errors. Risk assessment has been used extensively as a tool to address this type of uncertainty by estimating consequences of various assumptions about parameter values in the presence of uncertainty (Charles 1998). Decision analysis takes this one step further by weighting consequences estimated in risk assessments by the probabilities of their occurrences and
comparing the expected value of performance among various management options (Walters 1986). Alternatively, adaptive management can reduce uncertainties in parameter estimates either passively by collecting new information over time and updating parameter estimates accordingly, or actively by perturbing the system to investigate relations among variables (Walters 1986). However, changes in the structure of the system over time (e.g., the functional relation among variables) may confound efforts to learn about model parameters.

The third category of uncertainty, structural uncertainties, reflects ignorance about the nature of fisheries systems that results in incorrect assessments in population status and inappropriate harvest (or other management) policies to achieve objectives. Examples include uncertainty about the functional form of the relation between variables (e.g., between abundances of spawners and recruits), environmental influences on population dynamics, spatial structure of fish populations and harvesting, trends over time in biological and environmental parameters, management objectives, and harvesters' responses to fishing regulations. While managers have attempted to account for some forms of structural uncertainty using decision analysis, the potentially unlimited scope for this category of uncertainty can make quantitative analyses difficult. Charles (1998) suggests several ways to address this challenge: choosing management alternatives that are robust to key structural uncertainties, reducing those uncertainties using adaptive management, and guarding against irreversible negative consequences by using a precautionary approach to management.
Scales of uncertainties

Understanding the spatial and temporal scales at which uncertainties occur and interact can help managers and scientists identify and account for them. In fact, Levin (1992) argues that scale is the central problem in unifying basic and applied ecology and that much of the complexity of ecological systems arises from the interaction of processes occurring at different spatial and temporal scales. Moreover, Costanza et al. (1998) proposed that matching scales between biological and institutional systems is critical for effective fisheries management. For example, biological uncertainties in population abundances that persist at local scales (e.g., within a river system) may affect harvesters at the scale of management regulations (i.e., regional-scale), and may be affected by ocean-basin scale environmental changes. Therefore, regional-scale harvest policies should be responsive to variability at both stock and ocean-basin levels.

Hierarchy theory is frequently applied to problems in fisheries management because it provides a theoretical framework for relating processes at multiple scales. Systems are organized in hierarchical levels according to the scale at which processes occur while allowing for interactions with systems at lower and higher scales. For models of Pacific salmon (Oncorhynchus spp.) populations in the Columbia River basin, Washington, Lee and Grant (1995) proposed a hierarchical organization with three levels: a life-stage level to include uncertainties in spawning, rearing, and migration at seasonal time scales and local (stream-specific) spatial scales; a population level to include uncertainties in population dynamics at multiple-generation time scales and river-specific spatial scales; and an inter-population level to include uncertainties in long-term population survival at basin-wide spatial scales due to anthropogenic activities such as
logging impacts. In addition, Peterman (2004) suggested using hierarchical models as a tool to produce defensible informative priors for Bayesian analysis through the use of large data sets on multiple populations. By "borrowing" information from neighbouring populations (i.e., assuming some underlying structure at the metapopulation scale), it is possible to infer probability distributions for parameter values from populations for which data are poor or absent (e.g., Su et al. 2004). Furthermore, de la Mare (2005) proposed a hierarchical control system for marine management, where control systems occur at each hierarchical level and are linked by inputs (objectives) and outputs (attributes under control). This approach allows each part of the system (e.g., species or fishery) to be controlled locally, leaving supervisory control to higher multi-species or international levels.

Despite recent efforts to account for uncertainties at various temporal and spatial scales, and the interactions between them, many uncertainties remain that are ignored when evaluating management strategies. Specifically, uncertainties at long temporal scales and large spatial scales are often undetected because of the lack of power in large-scale, long-term monitoring programs (Maxwell and Jennings 2005), or they are deemed unimportant when making seasonal or annual decisions about harvest policies (Steele 1998).

**Outline of this thesis**

In this dissertation, I quantify sources and scales of uncertainties that have not been considered in previous efforts to model Pacific salmon fisheries, and I develop quantitative methods to account for them. Specifically, I focus on uncertainties in four aspects of fisheries management for Pacific salmon: (1) pre-season forecasting models
(i.e., models that forecast the abundance of adults vulnerable to the fishery), (2) outcomes of fishing regulations, (3) ability of management agencies to achieve stated objectives, and (4) ability to achieve collective action during international negotiations.

In the first chapter, I focus on parameter uncertainty of one type of forecasting model, a sibling model. That uncertainty results from observed long-term changes in the distribution of ages-at-maturity. Specifically, I address the following three questions:

1. Do long-term trends exist in parameters of the sibling model used to forecast recruitment of sockeye salmon (*O. nerka*) to fisheries in British Columbia and Alaska?

2. Can a forecasting model that tracks long-term trends in parameters forecast recruitment with lower mean-squared errors and with less bias than standard models, when evaluated in retrospective analyses?

3. What is the spatial scale of these trends, if they exist, and do they match those of potential environmental and biological drivers?

I analyze changes in parameters of the sibling model over a longer temporal scale and larger-spatial scale (across different regions in British Columbia and Alaska) than in previous studies.

In Chapter 2, I investigate the implications of uncertainty in outcomes (i.e., effects) of fishing regulations by answering the following two questions:

1. What is the magnitude and nature of deviations between management targets and the actual outcomes of regulations (outcome uncertainty) for sockeye salmon fisheries on the Fraser River, British Columbia?
(2) How can managers account for those patterns in outcome uncertainty when designing harvest policies?

Specifically, I assess both random interannual fluctuations in outcomes of fishing regulations and the structural form of these fluctuations (i.e., patterns in bias). I further examine variability in outcome uncertainties across management units (that comprise aggregates of populations) and compare these differences to environmental and management factors in order to improve our understanding of the sources of those uncertainties.

Although the modifications to current practices that I develop in the first two chapters may improve management performance by increasing managers' ability to control key attributes of the system (e.g., catch and abundance of spawners), the magnitude of these improvements is unknown, and it is not even clear that these improvement will occur. In fact, the "dual effect of control" (Walters 1986) may diminish contrast in abundances (range of observations), resulting in less-accurate parameters estimates in models used for stock assessment and management. In turn, this may reduce managers' ability to achieve stated objectives. In the third chapter, I use a simulation model to determine whether fishery management is improved by modifying current practices so that they account for long-term trends in age-at-maturity and forecasting uncertainty (Chapter 1) and outcome uncertainty (Chapter 2) once the remaining major components of the fishery system (and their associated uncertainties) are included.

In the fourth chapter, I examine implications of uncertainty for the outcome of negotiations for collective action to reduce competition by salmon from different countries that depend on a common pool of resources in the North Pacific Ocean. In
particular, evidence suggests that competition among salmon results in declines in age-specific body size of salmon (Peterman 1984; Ruggerone et al. 2003; Ruggerone and Nielsen 2004), which may reduce the total biomass and economic value of commercial catch. Although previous studies have examined the ocean-basin spatial scale and multi-decadal temporal scale of this problem from a biological perspective (e.g., Ishida et al. 1993; Ruggerone et al. 2005), none has extended the analysis to also address determinants of behaviour of nation states when negotiating cooperative agreements for this problem. Specifically, this chapter includes answers to the following questions:

(1) What are the incentives and disincentives for international cooperation in resolving the problem of competition among salmon in the North Pacific Ocean?

(2) What are potential strategies to alter the incentive structure to further encourage cooperation?

Finally, in the concluding section, I synthesize these chapters, outline several remaining challenges for fisheries management that this work reveals, and offer directions for future management efforts to overcome these challenges.

References


Chapter 1
Long-term Trends in Age-specific Recruitment of Sockeye Salmon (*Oncorhynchus nerka*) in a Changing Environment

Abstract

Sibling-age-class ("sibling") models, which relate abundance of one age class of adult sockeye salmon (*Oncorhynchus nerka*) to abundance of the previous age class in the previous year, are commonly used to forecast abundance one year ahead. Standard sibling models assume constant parameters over time. However, many sockeye salmon populations have shown temporal changes in age-at-maturity. We therefore developed a new Kalman filter sibling model that allowed for time-varying parameters. We found considerable evidence for long-term trends in parameters of sibling models for 24 sockeye salmon stocks in British Columbia and Alaska; most trends reflected increasing age at maturity. In a retrospective analysis, the Kalman filter forecasting models reduced mean-squared forecasting errors compared to standard sibling models in 29-39% of the stocks, depending on the age-class. The Kalman filter models also had mean percent biases closer to zero than the standard models for 54-94% of the stocks. Parameters of these sibling models are positively correlated among stocks from different regions, suggesting that large-scale factors (e.g., competition among stocks for limited marine prey) may be important drivers of long-term changes in age-at-maturity schedules in sockeye salmon.

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1 This chapter was published in the Canadian Journal of Fisheries and Aquatic Sciences (2004, 61: 2455-2470), co-authored with R.M. Peterman.
Introduction

Forecasting models are used widely in management of Pacific salmon (Oncorhynchus spp.) fisheries; good forecasts improve economic gains while reducing the chance of conservation problems. Inaccuracies in forecasts can result in either short-term economic costs from reduced harvest if too few fish are forecasted compared to actual returns, or conservation risks if forecasts are much higher than actual returns and result in harvests that are too high. To improve forecasting accuracy, we revised one type of forecasting model, the sibling-age-class relation (henceforth called the sibling model), and compared the performance of this new version with the standard sibling model.

Sibling models take advantage of the life history of Pacific salmon by forecasting the abundance of one age group based on its siblings' abundance, i.e., abundance of the previous age class that was estimated in the previous year. In sockeye salmon (O. nerka), for example, siblings, or offspring from a given brood class that were spawned in a given year, mature at various ages (from age three years to as old as age seven in some stocks). For instance, a sibling model can forecast the abundance of five-year-old sockeye recruits from the abundance of its four-year-old siblings that were estimated in the previous year as they returned to coastal waters.

Such sibling models are attractive because fish in the age group represented by the independent variable share similar environmental conditions with fish in the forecasted age group. Specifically, siblings that smolt in the same year (i.e., after either one or two winters in fresh water) but that mature and return to fresh water at different ages, experience similar conditions during early ocean residence, which is the most critical period for marine mortality (Groot and Margolis 1991). Therefore, siblings that
return to fresh water in successive years typically have similar relative variation in
recruitment over time. For example, abundance of age-four recruits to Chignik Lake,
Alaska that spend one winter in freshwater and two winters in the ocean (denoted age-
1.2), correlates well with abundance of sibling age-five recruits from the same brood year
that also spend one winter in freshwater but three winters in the ocean (denoted age-1.3)
(Fig. 1.1).

Sibling models are used widely for forecasting sockeye salmon recruitment in
British Columbia (B.C.) (Rutherford and Wood 2000) and Alaska (Cross and Gray 1999;
Hilborn et al. 1999). These models have the advantage of generating preseason forecasts
by using more recent stock information (age-specific returns from the previous year) than
stock-recruitment forecasting models, which use spawner abundances three to seven
years prior to recruitment. Therefore, sibling forecasting models may better reflect recent
changes in recruitment patterns than traditional stock-recruitment forecasting models or
their variants that incorporate environmental covariates. Nevertheless, there is still
considerable variation in recruitment that is not explained by sibling models (e.g., Fig.
1.1).

These forecasts of recruitment may be imperfect in part because the main
assumption of the sibling model may not be valid. In particular, the slope and intercept
parameters of the model may vary over time due to long-term changes in the proportion
of fish that mature at different ages. For example, mean age-at-maturity (affecting
proportions of recruits of each age) has shown a common increasing trend among 31
sockeye salmon stocks in B.C. and Alaska from the 1960s to the 1990s (Pyper et al.
1999).
There are at least two reasons why temporal trends in parameters of sibling models have not been analyzed in depth before. First, fisheries scientists have typically attributed changes in the age-specific recruitment to random sources of variability, and thus have assumed constant parameters over time for their forecasting models. In some cases, year-to-year changes in age-specific recruitment have been attributed to changes in body size of recruits. For example, in years where the body size of recruits of younger siblings is small, a greater-than-normal proportion of older siblings is expected to return the following year (L. Fair, personal communication, Alaska Department of Fish and Game, 333 Raspberry Road, Anchorage, Alaska, 99518-1599). However, long-term trends have not been investigated for sibling models, even though they have been identified in parameters of other models of salmon population dynamics (e.g., productivity parameter of stock-recruitment models for sockeye salmon, Peterman et al. 2003). Second, only recently has there been an assessment of the reliability of various methods for estimating temporal trends, or even changes to new persistent levels, in parameters of salmon models. Using Monte Carlo simulations, Peterman et al. (2000) found that if stock productivity changes over time, a model that allows for variation in the $a$ parameter of the Ricker stock-recruitment model and that is estimated using a Kalman filter tracks these changes better than a model with constant parameters, even if the latter model is updated annually. Such models with time-varying parameters also performed better in terms of mean-squared error than standard models (Peterman et al. 2000). Although that analysis pertained to the time-varying productivity parameter in stock-recruitment relations, its results suggest that a similar Kalman filter approach could be
used to determine whether long-term changes are occurring in parameters of sibling models.

Our overall goal was to improve forecasts of sockeye salmon recruitment using a revised sibling model with time-varying parameters. We had three specific research objectives. (1) We used a Kalman filter estimation scheme in conjunction with a sibling model that assumed time-varying parameters to identify whether there are long-term trends or persistent changes in parameters of sibling models for B.C. and Alaskan sockeye salmon stocks, as opposed to just year-to-year variation. (2) We used historical data in a retrospective analysis to compare the reliability of forecasts from the model that allows for temporal changes in its parameters with forecasts from the standard sibling model that assumes constant parameters over time. (3) We also examined potential physical and biological drivers of changes in parameters of sibling models.

**Methods**

**Biological data**

We analyzed 41 previously compiled time series of age-specific recruitment data (catch and numbers of spawners, or escapement) from a total of 24 sockeye salmon stocks and stock complexes (i.e., aggregations of salmon populations that return to the same river) from eight different management regions in B.C. and Alaska (Table 1; Fig. 1.2). Methods used by governmental agencies to estimate catches involved sampling at ports and performing stock identification and ageing of fish using scales, otoliths, and other means. These methods are reasonably reliable. Methods for estimating escapement are less reliable and varied somewhat among stocks. Those methods included counts at
fences, foot surveys, mark-recapture studies, and aerial surveys. As elaborated later in the Discussion section, it is unlikely that errors in methods of estimating abundance caused the temporal trends observed here to be spurious. We analyzed recruitment data for both the relation between age-1.2 and age-1.3 recruits, or age-1.z stanza, and the relation between age-2.2 and age-2.3 recruits, or age-2.z stanza. We did not analyze other age stanzas because most stocks had too few data points for this type of analysis. To ensure sufficient sample sizes, we only included stocks with at least 10 consecutive years of age-specific recruitment data and substantial abundances for both x.2- and x.3 age-classes. The time series ranged in duration from 14 to 77 years.

**Statistical analysis**

**Standard sibling model**

The standard sibling model forecasts the abundance of recruits returning in year $y$ in age-class $x.i$ (spending $x$ winters in freshwater, $i$ winters in the ocean), $R_{x.i,y}$, from the abundance of recruits that returned the previous year, $y-I$, in the previous age-class, $x.i-I$, $R_{x.i-1,y-1}$, (i.e., from the same brood year) (Peterman 1982):

$$\log_e (R_{x.i,y}) = a + b \log_e (R_{x.i-1,y-1}) + \nu, \quad (1)$$

Parameters $a$ and $b$ are specific to each age stanza, and $\nu$ is a normally distributed random error term that reflects the multiplicative log-normal variation often found in marine survival rates of salmon (Peterman 1981), specific to each brood year, $t$ ($t = y-4$ for 4-year-old recruits, and $t = y-5$ for 5-year-old recruits). This standard sibling model, as well as its Kalman filter version described below, is merely a descriptive model of past
data; it does not require a biological understanding of the mechanisms causing changes in recruitment.

A useful index of changing age structure, the $a$ parameter of this sibling model, reflects the ratio of the abundance of recruits of age-$x.i$ in year $y$ to the abundance of recruits of age-$x.i-1$ in year $y-1$, the latter raised to the power $b$ (as is apparent when equation 1 is rearranged):

$$a = \log_e \left( \frac{R_{x,i,y}}{(R_{x,i-1,y-1})^b} \right) - v_i .$$

The $a$ parameter thus represents an index of age-at-maturity when comparing two age-classes. When the $b$ parameter is equal to 1, the $a$ parameter represents the logarithm of the ratio of recruitment from two successive age-classes in two successive return years (i.e., from the same brood year and ocean entry year). Higher values of $a$ reflect a higher proportion of fish maturing at the older age for that particular age stanza than expected from the abundance of the next-younger-aged siblings. This $a$ parameter is thus a more refined measure of shifts in adult recruitment across ages than the more commonly used mean age-at-maturity, which averages across several ages and obscures shifts between its composite pairs of ages. Mean age-at-maturity is also not very useful for forecasting abundance of recruits in a given year because those recruits are typically composed of two or more age classes from two or more brood years; each of those brood years will likely have a somewhat different age structure.
Kalman filter model

We estimated temporal trends in the $a$ parameter of sibling models using a Kalman filter model. The Kalman filter allowed us to estimate temporal trends in observed age-specific recruitment that were due to systematic process variation separately from random sources of variation that were independent of those trends (Chatfield 1996; de Valpine and Hastings 2002). The Kalman filter approach thus permitted annual estimation of effects of underlying processes influencing age-specific recruitment while reducing the confounding from random sources of variability independent of the trend. This Kalman filter technique has been used previously to estimate changes in productivity of sockeye salmon in B.C. and Alaska as described by parameters of stock-recruitment models (Peterman et al. 2003).

This Kalman filter alternative to the standard sibling model consisted of two components. The “observation equation” described, for example, the observed relation between age-$x.3$ recruitment, $R_{x.3}$, and age-$x.2$ recruitment, $R_{x.2}$, from the successive return years, $y$ and $y-1$, respectively:

$$\log_e (R_{x.3,y}) = a_t + b \log_e (R_{x.2,y-1}) + \nu_t.$$  

This is the same form as equation (1) except that $a_t$ values can vary by brood year, $t$; the $b$ parameter remains constant. The error term, $\nu_t$, is normally distributed with a variance, $\sigma_v^2$. The “system equation” describes how the $a_t$ parameter (Y-intercept) changes over time as follows:

$$a_t = a_{t-1} + \omega_t.$$  


where $\omega_t$ is a normally distributed error term with a mean of zero and variance, $\sigma_\omega^2$. Here we assumed a random-walk process for the "system equation" because we had no a priori knowledge of temporal pattern in $a_t$ values. As well, Peterman et al. (2000) showed that a random-walk model performed well at tracking a wide variety of underlying temporal patterns in parameters. We estimated values for $a_t$ via a sequential updating procedure; the $b_t$, $\sigma_v^2$, and $\sigma_\omega^2$ were estimated via maximum likelihood (Chatfield 1996). For an in-depth description of methods of parameter estimation for a Kalman filter model, see Peterman et al. (2003).

Unlike equation (3), we originally investigated a Kalman filter in which both the $a$ and $b$ parameters (Y-intercept and slope) varied with time (i.e., $\log_e(R_{x,3y}) = a_t + b_t \log_e(R_{x,2y-1}) + v_t$). However, the estimated $b_t$ turned out to be constant for most stocks, whereas $a_t$ usually varied considerably with time. Furthermore, when data were subdivided into shorter time series (e.g., pre-1976 regime shift and post-1976 regime shift) and analyzed using the standard sibling model with constant parameters, the $a$ parameter differed between subsets, but the $b$ parameter did not. Therefore, we allowed only the $a$ parameter to vary in our analysis (equation 3).

For the Kalman filter model, we calculated annual values for $a_t$ (i.e., filtered values) using only data from the years prior to the year being estimated (Chatfield 1996; de Valpine and Hastings 2002). To estimate the best-fit values for $a_t$ over the entire data set after completing the estimation of all annual filtered $a_t$ values, we also calculated fixed-interval smoothed $a_t$ values (Harvey 1989), which used data from all years to estimate each $a_t$ value. The smoothed $a_t$ values were calculated sequentially backwards in time starting with the final year. The previous year’s smoothed estimate ($a_{t-1}$) was
calculated based on the smoothed and filtered estimates of $a_t$ from year $t$ and their variances. This procedure was repeated for each year in the data set. The filtered values reflected more short-term interannual variability in $a_t$ than the smoothed values. Because the smoothed $a_t$ estimates represent the optimal fit to the data (Chatfield 1996; de Valpine and Hastings 2002), we used these values in further analyses, unless stated otherwise. To permit comparisons among stocks with different mean ages at maturity, we standardized the time series of $a_t$ estimates for each stock and age stanza, using each stock’s respective mean $a_t$ value and variance.

**Model evaluation: retrospective analysis**

We used a retrospective analysis to compare the forecasts of the standard sibling model and its Kalman filter version. This method evaluates how models would have performed in actual practice; it uses only a subset of the data (e.g., the first $n$ years) to estimate model parameters and forecast the dependent variable in year $y+1$. This procedure is repeated iteratively for each year of data, using knowledge of only previous years’ data to estimate model parameters and make forecasts. Other approaches (e.g., cross-validation and bootstrapping, Shao 1993) were either difficult or inappropriate to use in this case due to the sequential nature of calculations for fitting the Kalman filter model; furthermore, their performance with Kalman filter models has not been evaluated.

Forecasts of abundance of recruits were calculated from the natural logarithm of abundances (equation 3) using the standard adjustment for back-transformation bias:

\[ R_{pred} = R_{non-adjusted} \cdot \exp(\sigma^2/2) , \]
where $R_{\text{pred}}$ is predicted recruitment after bias correction, $R_{\text{non-adjusted}}$ is the predicted recruitment prior to bias correction (e.g., $\exp(\log R_{x,y})$), and $\sigma^2$ is the residual variance of the forecasting model (Beauchamp and Olson 1973). Our two performance measures for comparing the standard sibling and Kalman filter models were: (1) mean-squared error, MSE (i.e., $\sum(R_{\text{pred}} - R_{\text{obs}})^2/N$, where $R_{\text{obs}}$ is observed recruitment and $N$ is the number of years of forecasted recruits), and (2) mean percent bias summed over each time series (i.e., $((\sum R_{\text{pred}} - \sum R_{\text{obs}})/\sum R_{\text{obs}}) \times 100$). We used the latter measure of bias instead of the commonly used annual mean percent error because the annual mean percent error has a positive bias when the observed recruitment has a large random component. For example, when the coefficient of variation of the residual forecasting error is 0.5, the mean percent error calculated from each observation separately will be positively biased by 25% (B. Pyper, personal communication, S.P. Cramer & Associates Inc., 600 NW Fariss Road, Gresham, Oregon, 97030).

**Spatial covariation among stocks in the $a_i$ parameter of the sibling model**

We identified the spatial scale of physical and biological processes driving changes in $a_i$ by estimating the spatial scale across which $a_i$ values were positively correlated among stocks. We examined four hypotheses for the spatial scale of these processes. First, local-scale (i.e., stock-specific) mechanisms may be responsible for variations in $a_i$, resulting in no correlation of $a_i$ among stocks. Second, regional-scale mechanisms may be responsible for variations in $a_i$, resulting in correlations among stocks within regions (e.g., Bristol Bay, Alaska), but not among two or more of our eight regions (e.g., between Bristol Bay and Kodiak, Alaska). Third, large-scale mechanisms may be responsible for variations in $a_i$, resulting in correlations among stocks within a
region as well as among regions. For this hypothesis, we defined large-scale as the spatial scale at which stocks from several regions can be influenced simultaneously (e.g., the area of overlap in ocean distribution for sockeye stocks from across B.C. and Alaska, French et al. 1976). Fourth, a combination of these mechanisms may act at various scales, resulting in different correlations within and among regions.

To examine these hypotheses, Pearson product-moment correlation coefficients were calculated between the \( a_t \) series for each pair of stocks using the maximum number of years possible for each paired comparison. Mean correlation coefficients were calculated both within and among regions using Fisher’s z-transform of each correlation coefficient to normalize the distribution prior to calculating the mean (Zar 1999). We were interested in the general overall pattern of correlations among stocks within and among regions, rather than in particular pairwise comparisons.

**Correlations with physical and biological variable**

Using correlation analysis, we then explored physical and biological factors that might be indirectly associated with changes in the \( a_t \) parameter of sibling models through changes in oceanographic conditions that affect body growth rates, for instance. Unfortunately, there is no long-term data series for marine food supply for salmon or size-selective predation over the decades covered by our data to test detailed hypotheses about environmental mechanisms; Ocean Station P zooplankton data were collected only up through 1980. We therefore used various surrogates of ocean conditions. Because we found evidence for trends in \( a_t \) values correlated among stocks over wide areas (see Results below), we only examined physical and biological factors that exhibit positive covariation across reasonably large spatial scales. This reduced the chance of spurious
correlations between the $a_i$ parameter of sibling models and physical and biological factors.

The first large-scale physical factor was winter mean Pacific Decadal Oscillation (PDO), an index of ocean-basin-wide changes in climate (Mantua et al. 1997) (available from the Joint Institute for the Study of Atmosphere and Ocean, University of Washington, Seattle, URL: http://jisao.washington.edu/pdo/PDO.latest). The PDO is the first principle component of the mean monthly sea-surface temperature (SST) anomalies across the North Pacific Ocean poleward of 20°N (Mantua et al. 1997); those authors suggest that positive deviations in PDO correlate with increased phytoplankton and zooplankton productivity. The second physical factor was deviation from the winter mean SST in the region where sockeye salmon from different stocks in the Northeastern Pacific overlap in the Gulf of Alaska, which was an area enclosing the line of 45°N from 170°W to 150°W in the south, the line of 55°N from 155°W to 140°W in the north, the point 50°N and 175°W in the west, and the point 50°N and 145°W in the east (Pyper and Peterman 1999). We used monthly SST (°C) on a 5°×5° longitude-latitude grid across this area (D. Cayan and L. Riddle, personal communication, Climate Research Division, 9500 Gilman Dr., Scripps Institute of Oceanography, La Jolla, California, 92093; i.e., the same data as Pyper and Peterman 1999). Mean winter SST deviations were calculated by subtracting the long-term mean SST for 1947 to 1997 from the SST for each grid cell and month, and then averaging across winter months (November through February). SST may directly reflect changes in temperature-dependent physiological processes. Winter SST may also indirectly indicate ocean conditions associated with altered prey or predator abundances and distributions because climatic forcing during the winter
influences the subsequent spring and summer ocean productivity (Brodeur and Ware 1992).

In addition to physical factors, biological factors that affect or reflect altered growth rates of body size have been used to explain age-at-maturity schedules (Bigler et al. 1996; Pyper et al. 1999). Therefore, we also estimated correlations between sibling-model parameters and the following biological factors: (1) the natural logarithm of total abundance of sockeye salmon recruits in all B.C. and Alaska stocks; (2) the natural logarithm of abundance of Bristol Bay sockeye salmon recruits (used for a separate analysis of Bristol Bay stocks only, because these stocks have common migratory patterns and may experience more competition from stocks within Bristol Bay than from stocks from other regions in B.C. and Alaska); and (3) the mean body size of age-x.2 female sockeye salmon recruits, which were only available for Bristol Bay stocks (Pyper et al. 1999). The first two factors, abundances of conspecifics, reflect competition for limited food supplies (Bigler et al. 1996; Kaeriyama 1998), which affects body size and growth-dependent maturity schedules. For example, Peterman (1984) estimated that age-specific body weight at maturity for sockeye salmon in B.C. decreased by 10-22% when abundance of conspecifics was high during early ocean life. We used natural logarithms of abundances, which normalized residual errors for regressions. The third factor, body size of recruits, may reflect competition due to density-dependent growth in the ocean. Body size during the penultimate year of ocean life when the maturation process is initiated was not available from back-calculating from scales or any other source. We therefore used age- and stock-specific body size at maturity as an indirect index of body size during the prior ocean residence period. We used the mean mid-eye-to-fork length
of age-x.2 female spawners (Pyper et al. 1999) because body sizes of age-x.3 female spawners in year y are of course unknown when forecasting age-x.3 recruitment in year y.

In the correlation analysis, time-series of sibling-model $a_t$ values were aligned with data on physical and biological variables at a variety of time lags, depending on various hypotheses about the period of ocean residence when these variables are most critical for salmon growth and maturation. Environmentally induced changes in age-at-maturity appear to occur within the first two years of ocean residence (Peterman 1985; Pyper et al. 1999). However, marine growth, which is associated with subsequent age-at-maturity (Bigler et al. 1996; Pyper et al. 1999), appears to be influenced by environmental conditions at various ocean ages. For instance, Rogers and Ruggerone (1993) found that growth increments from scale analyses were correlated with environmental conditions during the first and second years of ocean life for one sockeye salmon stock (Nushagak, Alaska). In contrast, LaLanne (1971) and Pyper et al. (1999) suggest that conditions nearer to the time of return migration have the largest influence on growth and final adult body size (penultimate and ultimate year of ocean life). For fish spending 2 winters in the ocean (age-x.2), these periods suggested by different authors will overlap (e.g., 2nd year of ocean residence is equivalent to the penultimate year). Furthermore, Ruggerone et al. (2003) found that density-dependent interactions between Alaskan pink (*O. gorbuscha*) and sockeye salmon during the second and third years of ocean life (OEY+1 and OEY+2, where OEY is ocean-entry year) were critical in determining body size at maturity. Therefore, we first aligned $a_t$ series from sibling models with physical conditions and abundances of sockeye salmon recruits that returned during the second year of ocean life (i.e., the abundance of recruits in the penultimate
year for age-x.2 fish). We also examined correlations between \(a_t\) values and both physical factors and abundances of sockeye salmon recruits aligned by earlier ages (OEY) and later ages (OEY+2 and OEY+3) to include factors influencing age-at-maturity and growth at different periods of ocean life. Only a small proportion of the total abundance of recruits return after more than three winters in the ocean (average of < 1% for stocks considered here); therefore, older age-classes and later alignments were not considered.

Autocorrelations within each time series of explanatory variables and \(a_t\) values increased the chance of incorrectly finding a significant correlation. To reduce this bias, we used the modified Chelton method suggested by Pyper and Peterman (1998) to adjust the degrees of freedom for the hypothesis tests in correlation analyses. Although autocorrelations of some series of \(a_t\) values were larger than the range of time series previously simulated by Pyper and Peterman (1998) (> 0.9), biases in type I error rates were low for this method even when autocorrelation was 0.9 (the simulated type I error rates were <0.07 when the true type I error was 0.05). Hypothesis testing in the correlation analysis with environmental variables was appropriate because the assumption that variables are bivariate-normally distributed was met.

We calculated the probability, \(P\), of finding the observed number of significant correlations between \(a_t\) values and explanatory variables by chance from a binomial distribution, and compared this to the number of significant correlations observed at \(\alpha = 0.05\). Only those stocks for which \(a_t\) varied over time were included in the analysis that explored various explanatory variables.
Multiple regression

The combined influence of physical factors (SST and PDO) and biological factors (sockeye salmon abundances and age-specific body size at maturity) on temporal variability in \( a_i \) was estimated in multiple regressions for each stock individually:

\[
a_{it} = c_{i0} + c_{i1} (E_1)_{t+x+1} + c_{i2} (E_2)_{t+x+1} + \ldots + \varepsilon_{ij},
\]

where \( E_1, E_2, \ldots \) are physical and biological variables experienced during ocean residence in year \( t+x+1 \) (OYE +1), \( t \) is brood year, \( x \) is the freshwater age, \( c_{i0}, c_{i1}, c_{i2} \ldots \) are constants for each stock, \( i, \) and \( \varepsilon_{ij} \) is a normally distributed error term with variance, \( \sigma^2. \)

The assumptions of normality and constant variance were met in this regression model, but the errors were not independent of each other. Therefore, we also separately fit another multiple regression model that accounted for autocorrelation in the residuals of model (6) using an autoregressive term of lag 1 (i.e., an AR(1) model). We did this because Bence (1995) showed that if such autocorrelation is ignored, confidence intervals around the parameters may be underestimated, resulting in type I errors (i.e., including a parameter in the model when the coefficient is not actually significantly different from zero). The model form was the same as the standard multiple linear regression model (equation 6) except the error term, \( \varepsilon_i \), took the form:

\[
\varepsilon_i = \rho \varepsilon_{i-1} + \Phi_i,
\]

where \( \varepsilon_i \) are autocorrelated residuals of the linear regression, \( \rho \) is the lag-1 autoregressive coefficient, and \( \Phi_i \) is a normally distributed random-error term.

We used stepwise regressions to add environmental variables to the models (equation 6) that could explain a significant amount of additional variance in \( a_i \) beyond
that explained by previously selected variables for each stock. There was no evidence for curvilinear relations between $a_i$ values and environmental variables.

**Results**

**Kalman filter**

The majority of sockeye salmon stocks that we examined showed evidence for substantial increasing temporal trends in the $a_i$ values of sibling models (e.g., Naknek River, Alaska, and Skeena River, B.C. stocks, Fig. 1.3). Temporal variability existed in the filtered $a_i$ values for all time series, and most stocks also showed variability over time in the smoothed $a_i$ values (20 out of 24 for the age-1 stanza -- Fig. 1.4; 13 out of 18 for age-2 stanza -- Fig. 1.5). A few stocks and age stanzas showed no trend in smoothed $a_i$ (horizontal lines in Figs. 1.4 and 1.5), because in those cases all of the variance was attributed to observation error by the Kalman filter’s estimation procedure, and none to the systematic process error. Smoothed $a_i$ values were larger than filtered $a_i$ values for a given brood year for many stocks (e.g., Fig. 1.3) because smoothed estimates, $a_i$, were based on the weighted average of the smoothed estimate $a_{i+1}$, and the filtered estimate $a_i$. When filtered $a_i$ increased over time, as was typically the case, this tended to produce smoothed values that were larger than filtered values. By showing the smoothed $a_i$ values in standardized units (mean equal to zero and standard deviation equal to one) and grouping them according to region (Figs. 1.4 and 1.5), the similarity in increasing trends over time is evident for most stocks. These temporal trends reflect increasing numbers of fish maturing at later ages for a given abundance maturing at the preceding age.
Model evaluation: retrospective analysis

Our retrospective analyses showed that although the standard sibling model was still best in most stocks, there were many instances in which the Kalman filter model improved forecasts of recruits. Specifically, in 29% of the stocks for the age-1.z stanza and 39% for age-2.z, the Kalman filter model (which assumed time-varying $a_z$ values) had a smaller mean-squared error (MSE) of forecasts than the standard sibling model (which assumed a constant $a_z$). That is, the ratio of the MSE$_{KF}$ to MSE$_{standard}$ was less than 1.0 for these particular stocks (Fig. 1.6). Nevertheless, the MSEs of the Kalman filter models were much greater than those of the standard sibling models in many other stocks (Fig. 1.6).

The ratios of the MSEs from the Kalman filter models to the MSEs from the standard sibling models differed among regions. For example, these ratios were lower for Bristol Bay stocks compared to B.C. stocks and the other Alaska stocks (Fig. 1.6). Among Bristol Bay stocks, the MSEs from the Kalman filter models were lower than the MSEs from the standard sibling models for 63% of stocks for the age-1.z stanza and 38% of stocks for age-2.z. Time series were also longer for the Bristol Bay stocks compared to most stocks from other regions (Table 1.1). The duration of time series was negatively correlated with the ratios of the MSEs for the Kalman filter models to the MSEs of the standard sibling models, but only for the age-1.z stanza ($r^2 = 0.72$, $P < 0.01$ for age-1.z stanza; $r^2 = 0.28$, $P > 0.1$ for age-2.z). Stocks with longer time series were better fit by the Kalman filter models than the standard sibling models due to increases over time in forecasting errors for the standard models for some stocks.
Over the long term, the mean percent bias was closer to zero for the Kalman filter models than the standard sibling models for most stocks (54% for age-1.z stanza, 94% for age-2.z stanza) (Figs. 1.7 and 1.8, respectively). One stock, Nelson River, showed a substantially higher positive bias than the other stocks for both the Kalman filter and standard sibling models for the age-1.z stanza (Fig. 1.7) and for the standard sibling model for the age-2.z stanza (Fig. 1.8). For this stock, large forecasting errors occurred in 1987 due to high age-1.2 recruitment in 1986 followed by anomalously low age-1.3 recruitment in 1987. This one year of high forecasting error contributed strongly to the mean percent bias for that stock because of the short duration of the data set (14 years).

**Spatial covariation of the sibling-model \( a_r \) values among stocks**

The time series of smoothed \( a_r \) values showed strong positive correlations both within and among regions (Figs. 1.9 and 1.10). Positive covariation was apparent across both regional- and ocean-basin-scales, the latter indicated by correlations among B.C. and Bristol Bay, Alaska stocks, for example. The mean correlation coefficient was higher for comparisons of stocks within regions than for comparisons among regions for both the age-1.z (\( r_{within \ regions} = 0.85, \ r_{among \ regions} = 0.63 \)) and age-2.z (\( r_{within \ regions} = 0.85, \ r_{among \ regions} = 0.56 \)) cases. Correlations among smoothed \( a_r \) values may be inflated due to autocorrelation within each time-series, however, this cannot explain the observed differences in correlations within and among regions; the filtered \( a_r \) values showed similar patterns of covariation with somewhat lower magnitude for both age-1.z (\( r_{within \ regions} = 0.62, \ r_{among \ regions} = 0.40 \)) and age-2.z (\( r_{within \ regions} = 0.75, \ r_{among \ regions} = 0.50 \)) cases.
One stock, Black Lake, Alaska, showed considerable differences between its stock-specific trend in $a_t$ for the age-1.2 stanza and both the ocean-basin-scale and regional-scale patterns that were common to most other sockeye salmon stocks (Fig. 1.9). This trend in $a_t$ was highly negatively correlated ($r < -0.5$) with most other stocks. Note that $a_t$ values from Black Lake extend from 1918 to 1994 (top right, Fig. 1.4), though most correlations in Fig. 1.9 were computed for only the latter half of the time series (post-1950) when recruitment data for other stocks became available. In contrast to the increasing values observed in most stocks since 1965, $a_t$ values decreased for Black Lake during this period.

Compared to the standard sibling models, the Kalman filter models were better able to distinguish trends in the natural logarithm of age-specific recruitment that are shared among stocks within and among regions. The time-varying estimates of smoothed $a_t$ from the Kalman filter model showed stronger covariation among stocks, both within regions and among regions, than the residuals from the standard sibling model (equation 1), for both the age-1.2 and age-2.2 stanzas (Fig. 1.11). This suggests that $a_t$ values capture more of the systematic trends in the sibling relation shared among stocks than residuals from the standard sibling model.

**Correlations with physical and biological variables**

We found a preponderance of significant positive correlations between $a_t$ values and each of PDO, SST in the Gulf of Alaska, total sockeye salmon abundances, and Bristol Bay sockeye salmon abundances when time series were aligned by OEFY+1 (Table 1.2). These positive correlations, which were significant after taking autocorrelation into account, were found in 61% of the analyses of pairs of time series for age 1.2 and 59%
for age 2.2, many more than would be expected by chance alone if the correlations were in fact zero (Table 1.2). In particular, as PDO, SST, total sockeye salmon abundances, and Bristol Bay sockeye abundances increased, a larger number of fish tended to return for a given recruitment at the preceding age in the previous year (i.e., \( a_i \) increased); this was true for both age stanzas (Table 1.2). Higher values of the physical variables (PDO and SST) represent higher temperatures and also indirectly indicate higher phytoplankton and zooplankton productivity (Mantua et al. 1997). The biological variables, total and Bristol Bay sockeye abundances, are an index of competition among stocks. Negative correlations were infrequent for the correlations between \( a_i \) and both the physical and biological variables mentioned above (PDO, SST, and sockeye abundances) (Table 1.2). Similar significant positive correlations were found for PDO, SST, and sockeye abundances when the alignment of data series was changed to OYE, OEY+2, or OEY+3. One exception was that most correlations between PDO and parameters of sibling models were not statistically significant (\( P > 0.05 \)) when aligned by OEY+3 (i.e., fewer cases were significant than expected by chance alone).

In contrast, we found significant negative correlations between \( a_i \) and body size of age-1.2 recruits in three of seven Bristol Bay stocks for the age-1.2 stanza (Table 1.2, bottom). For that age stanza, there were more significant negative correlations than expected by chance alone if there were no correlation (\( P < 0.001 \)). In other words, as the body size of age-1.2 recruits declined, a larger number of fish tended to return at age-1.3 for a given abundance at age 1.2. There were no significant positive correlations for these comparisons. For fish that spent two winters in freshwater (age-2.2 stanza), no significant correlations existed between \( a_i \) values and body size of age-2.2 recruits (Table
1.2, bottom). The exclusion of some stocks due to stationary $a_t$ values probably did not bias our results because only a small number of stocks were excluded from this analysis (four out of 24 and five out of 18 stocks for age-1.z and age-2.z stanzas, respectively).

**Multiple regression**

To obtain further information on the influence of each environmental variable on $a_t$, we performed a multiple linear regression aligning data for independent variables by OEy+1. The natural logarithm of total sockeye salmon abundance explained a significant portion of the variability in $a_t$ in more time series than any other environmental variable (29 out of 33 time series) (Table 1.3). When modelling $a_t$ values as a function of physical and biological variables, SST was included as an explanatory variable in almost half of these cases (17 out of 33). PDO and Bristol Bay sockeye salmon abundances were only included in a small number of models, and therefore could explain variability in $a_t$ beyond that explained by total sockeye salmon abundance and SST in only a small portion of the stocks. However, multicollinearity among some of these environmental variables (Table 1.4) creates confounding of interpretation of these results. More importantly, autocorrelation in the residuals (Table 1.3) prompted us to also fit a multiple regression (equation 6) that included an autoregressive lag-1-year error term (equation 7). Once that autocorrelation was accounted for, none of the environmental variables was consistently significant in the model, often resulting in the null model (i.e., a model containing the Y-intercept only).
Discussion

Many North American sockeye salmon stocks show increases in $a_i$ parameters of their sibling models over the last 40 years. These changes reflect increases in the number of older fish maturing and recruiting relative to younger ones from the same brood-class and ocean-entry year. In other words, more adults have returned recently at ages 1.3 and 2.3 for a given abundance of recruits of ages 1.2 and 2.2, respectively. For example, for the age-1.2-to-1.3 stanza for the Kvichak stock (Bristol Bay, Alaska), the $a_i$ values increased from 3.5 in 1952 to 4.3 in 1996, reflecting a change in the ratio of age-1.3 to age-1.2 recruitment from 0.7:1 in 1952, to 3.2:1 in 1996. These increases in $a_i$ values were qualitatively similar to increases in mean age-at-maturity over the same time period for 31 sockeye salmon stocks in British Columbia and Alaska (Pyper et al. 1999). Only a few stocks showed little or no evidence of consistent temporal trends in $a_i$ over time.

These results of increasing values of $a_i$ over time for most stocks are unlikely to be a spurious result of errors in estimating escapements. For us to have come to the wrong conclusion about increasing trends in sibling model $a_i$ values, there would have to have been a systematic time trend toward increasingly overestimating the proportion of older fish relative to younger ones in each successive year.

Incorporating temporal variability in the Y-intercept ($a_i$) of the sibling models using a Kalman filter resulted in retrospective forecasts with mean percent biases closer to zero for most stocks compared to using standard sibling models, which assume constant parameters over time. However, the Kalman filter model did not improve the mean-squared error of forecasts of recruitment consistently across all stocks that showed changes in age-specific recruitment over time. Kalman filter models reduced the residual
errors from retrospective forecasts for only 29% of the stocks for the age-1.z stanza and 39% of the stocks for age-2.z, compared to standard sibling models. Nevertheless, Kalman filter models improved forecasts for a higher proportion of Bristol Bay stocks than for the other Alaskan plus B.C. stocks. This was especially true for the measure of bias of forecasts. It is not clear whether this result for Bristol Bay stocks was due to (1) a larger change in the $a_i$ parameters in that region, (2) the longer average duration of Bristol Bay sockeye time series compared to other stocks, both of which might favour parameter estimation with the more complex Kalman filter model, or (3) a combination of these factors. Nonetheless, Kalman filter models should be seriously considered for forecasting age-specific recruitment in at least the Bristol Bay region; other regions may or may not benefit from this model in the future as more data accumulate.

The numerous cases of positively biased forecasts from the Kalman filter model, especially for stocks outside of Bristol Bay, Alaska, can be explained by the observation that when age-x.2 recruitment was high, this model occasionally greatly overestimated age-x.3 recruitment in the subsequent year. This phenomenon was likely due to the annually updated estimates of $a_i$ being substantially altered due to large outliers in the data. In such cases, the changing $a_i$ may tend to follow the noise in the sibling relation rather than the systematic changes that we hoped to capture.

In addition to improving forecasts for some stocks, Kalman filter models, when compared with standard sibling models, can help to better identify systematic trends in age structure of recruits that are associated with long-term physical and biological processes. Kalman filters do this by reducing the confounding of interpretation resulting from random sources of error independent of these trends. These less-confounded
estimates of systematic process variation reflected by $a_i$ values can lead to higher-power tests of mechanisms causing changes in proportional age distributions of sockeye salmon recruits.

Further information about the type of environmental processes that strongly affect changes in age-at-maturity is provided by the relative magnitude of pairwise correlations of $a_i$ series at the regional and ocean-basin scales. The mean correlation of stock-specific smoothed $a_i$ values among stocks at the regional scale (0.85 for both age stanzas) was positive and greater than the mean correlation among stocks at the ocean-basin scale (0.56 to 0.63, depending on the age stanza). These mean correlations suggest that processes that are most important for changes in stock-specific $a_i$ values are shared among stocks at both regional and ocean-basin scales. The larger mean correlation within regions can be explained by a combination of regional- and ocean-basin-scale processes, whereas the mean correlation among regions reflects only ocean-basin scale processes.

In contrast, local-scale processes may dominate for a few stocks such as Black Lake, overriding regional or large-scale processes. In that stock, the age-1.2 $a_i$ parameter declined after the 1970s as a result of declines in age-1.3 recruitment, without concurrent declines in age-1.2 recruitment. The unique hydrogeological characteristics of Black Lake may explain its decreasing $a_i$ values over the last 30 years, which are unlike the increasing $a_i$ values generally shown by other stocks in that region. Since the 1960s, Black Lake has experienced substantial reductions in water volume because of erosion of outflow rivers (Ruggerone et al. 2001). Lower water levels have reduced in-lake habitat area and have made upstream migration more arduous, especially for larger adults.
(Ruggerone et al. 2001). Therefore, larger (and older) recruits may experience higher pre-spawning mortality before they are enumerated, thereby resulting in fewer age-1.3 recruits compared to age-1.2 recruits and decreased $a_t$ values.

Pyper et al.'s (1999) analysis of the spatial scale of covariation in mean age-at-maturity concluded that ocean-basin-scale processes dominate over those acting on age-at-maturity at regional scales. However, in addition, Pyper et al. (1999) found substantial stock-specific variation in mean age-at-maturity, whereas we found that stock-specific variations in $a_t$ were small compared to regional and large-scale variations. The difference between the results of these two studies can be explained by the different metrics used. First, the mean age-at-maturity used by Pyper et al. (1999) is a weighted average that integrates recruitment over all age classes from brood classes that entered the ocean in the same year (e.g., age-1.1, 1.2, 1.3, and 1.4 classes). In contrast, our sibling-model $a_t$ parameter reflects the association between the logarithm of abundances of only two age-classes (e.g., age-1.2 and age-1.3). Therefore, differences at younger (age-1.1) and older (age-1.4) age-classes will not be represented in the $a_t$ values estimated here. Second, mean age-at-maturity is calculated for each brood year and is sensitive to annual anomalies such as extremely high recruitments of specific age classes, whereas $a_t$ values are calculated using data from all years and reflect longer-term trends in age structure that are normally masked by random interannual variability. Therefore, in addition to long-term trends, mean age-at-maturity will be more likely to reflect patterns at short temporal scales that are often associated with patterns at small spatial scales (Holling 1992) and that are not detected by sibling-model $a_t$ values.
Our results suggest that processes acting at regional, as well as ocean-basin-spatial scales, are mostly responsible for the temporal increases in sibling-model $a_i$ values. The ocean-basin-scale processes may influence fish from different stocks either when they co-occur in the Gulf of Alaska or at other points in their ocean life when they still share large-scale physical or biological conditions. Alternatively, each stock may be independently influenced by separate conditions, which have coincidentally resulted in similar trends in $a_i$ values.

For at least two reasons, it is difficult to attribute particular causal mechanisms to the observed covariation in sibling-model parameters at either regional or ocean-basin scales. First, there is some multicollinearity among the physical and biological explanatory variables (Table 1.4), as well as autocorrelation in their time series, which precludes convincingly distinguishing the separate influences on sibling-model $a_i$ values of abundance of conspecific competitors, age-specific body size, or ocean temperatures (PDO and SST) and the conditions that the latter indirectly reflect. Specifically, in the multiple regressions, collinearity between total sockeye salmon abundance and PDO means that PDO cannot explain much additional variance in $a_i$ values not already explained by sockeye salmon abundances, and vice versa. Second, some unspecified latent variable(s) may directly affect both $a_i$ and one or more of our explanatory variables, coincidentally creating the observed correlations between the latter and $a_i$. For example, none of the environmental variables was consistently significant in our analyses that assumed a lag-1 autocorrelated residual variation. Third, but less problematic, correlations in $a_i$ values among stocks may cause an underestimate of the frequency of type I errors in the correlation analyses between $a_i$ values and explanatory variables (i.e.,
resulting in too many significant cases from P values that are too low). However, the P values were well below \( \alpha = 0.05 \) (\( P < 0.001 \) for most comparisons), so it is unlikely that this bias will change the interpretation of results.

Nevertheless, recall that the total abundance of sockeye salmon in the Gulf of Alaska was the variable we examined that was most frequently (and positively) associated with \( a_i \) in our multiple regressions (29 of 33 cases). If that abundance of conspecifics is actually the most important explanatory variable for the temporal increases in \( a_i \) values, this would be consistent with the large amount of previous research documenting the interaction among increased abundance of conspecifics, reduced density-dependent growth, and delayed maturation of sockeye salmon. For instance, an inverse relation between total abundances of chum salmon (\( O. keta \)) (or sockeye salmon) and growth during their marine life is well documented based on body size-at-maturity (Peterman 1984; Kaeriyama and Katsuyama 2001; Ishida et al. 2002), scale analyses (Rogers and Ruggerone 1993; Ruggerone et al. 2002), and stomach contents (Tadokoro et al. 1996; Davis et al. 1998). Such effects of a given abundance of competitors are, of course, influenced by oceanographic conditions that affect biological productivity of food for salmon (e.g., Aydin et al. 2000; Kaeriyama et al. 2000). The final link between abundance and \( a_i \) values is that relatively small age-specific body size at maturity tends to be associated with late age-at-maturity in several salmon species, including sockeye (Kaeriyama 1998; Pyper et al. 1999). Furthermore, sockeye salmon abundances have increased in the Gulf of Alaska over the previous 30 years (Pyper and Peterman 1999). Given the above linkages among abundance of competitors, growth rate, and age-at-
maturity in salmon, the increasing $a$, values over time for sibling models reported here are thus consistent with that temporal trend in abundance.

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**References**


Table 1.1 Summary of 24 sockeye salmon stocks (*Oncorhynchus nerka*) analyzed with sibling models.

"Region" and "Stock" describe a stock's location; "N" is the duration of the time series (years); "Age-class stanza" shows which age classes were the X and Y variables, respectively; "Lag(1)" is the one-year-lag autocorrelation coefficient in residuals of the standard sibling model that assumes constant parameters; an asterisk denotes a significant autocorrelation at the $\alpha = 0.05$ level; "Average recruitment" is the average recruitment over each time series of each age-class in thousands of fish; "Source" describes from whom the data were obtained. All sources are personal communications, showing current addresses in the footnote.

<table>
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<tr>
<th>Region</th>
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<th>N</th>
<th>Age-class stanza</th>
<th>lag 1</th>
<th>Average recruitment in 1000s (x.2, x.3)</th>
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<td>Age-class stanza</td>
<td>lag 1</td>
<td>Average recruitment in 1000s (x.2, x.3)</td>
<td>Source</td>
</tr>
<tr>
<td>----------------</td>
<td>------------</td>
<td>------------</td>
<td>----</td>
<td>------------------</td>
<td>-------</td>
<td>--------------------------------------</td>
<td>-------------------------</td>
</tr>
<tr>
<td>Nass River</td>
<td>1967-1992</td>
<td>26</td>
<td>1.2 vs. 1.3</td>
<td>0.12</td>
<td>(211, 150)</td>
<td>Les Jantz, Fisheries and Oceans Canada</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>25</td>
<td>2.2 vs. 2.3</td>
<td>0.36*</td>
<td>(248, 41)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Central</td>
<td>Long Lake</td>
<td>1973-1994</td>
<td>20</td>
<td>1.2 vs. 1.3</td>
<td>-0.29*</td>
<td>(107, 200)</td>
<td>Chris Wood, Fisheries and Oceans Canada</td>
</tr>
<tr>
<td>British</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Columbia</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Fraser</td>
<td>Pitt</td>
<td>1948-1994</td>
<td>47</td>
<td>1.2 vs. 1.3</td>
<td>-0.30*</td>
<td>(25, 45)</td>
<td>Jim Woodey and Mike LaPointe, Pacific Salmon Commission</td>
</tr>
<tr>
<td>River</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

1 The mailing addresses for the data sources are, in order of citation: Michael Link, LGL Alaska Research Associates, 1101 East 76th Ave., Anchorage, Alaska, 99518; Patricia Nelson, Alaska Dept. of Fish and Game, 1390 Buskin River Rd., Kodiak, Alaska, 99615; Ken Tarbox, Alaska Dept. of Fish and Game, 43961 Kalifornsky Beach Road, Suite B, Soldotna, Alaska, 99669-8367; Mark Willette, Alaska Dept. of Fish and Game, 43961 Kalifornsky Beach Road, Suite B, Soldotna, Alaska, 99669-8367; Chris Wood, Conservation Biology Section, Fisheries and Oceans Canada, Pacific Biological Station, 3190 Hammond Bay Road, Nanaimo, British Columbia, V9T 6N7; Les Jantz, Fisheries and Oceans Canada, 417-2nd Avenue West, Prince Rupert, British Columbia, V8J 1G8; Jim Woodey and Mike LaPointe, Pacific Salmon Commission, 600 - 1155 Robson St., Vancouver, British Columbia, V6E 1B5).
Table 1.2 Summary of correlations between smoothed Kalman filter $a_t$ values for the sibling model and time series of physical or biological variables.

"Physical or biological variable" and "Age-class stanza" describe the variables in each pairwise correlation for which $a_t$ values were used; age-class stanza 1.z and 2.z represent the relation between age-1.2 and age-1.3 fish, and between age-2.2 and age-2.3 fish, respectively. PDO is the Pacific Decadal Oscillation. SST is sea-surface temperature. "N" is the total number of correlations calculated (corresponding to the number of stocks analyzed). The number of significant positive correlations at the $\alpha = 0.05$ level, and the median and range for these correlations ($r$) are shown. $P(\cdot)$ is the probability of finding that number of significant positive correlations by chance alone, calculated from the binomial distribution. Similarly, we show the number of significant negative correlations at the $\alpha = 0.05$ level, the median and range for these correlations, and the probability of finding that number of significant negative correlations by chance alone, $P(\cdot)$.

<table>
<thead>
<tr>
<th>Physical or biological variable</th>
<th>Age-class stanza</th>
<th>N</th>
<th>Number of significant positive correlations</th>
<th>Median $r$ for significant positive correlations (range)</th>
<th>$P(\cdot)$</th>
<th>Number of significant negative correlations</th>
<th>Median $r$ for significant negative correlations (range)</th>
<th>$P(\cdot)$</th>
</tr>
</thead>
<tbody>
<tr>
<td>$\log_e$(total sockeye abundance)</td>
<td>1.z</td>
<td>20</td>
<td>17</td>
<td>0.65</td>
<td>$&lt; 0.001$</td>
<td>1</td>
<td>-0.64</td>
<td>$&gt; 0.2$</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>(0.54, 0.76)</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>2.z</td>
<td>13</td>
<td>10</td>
<td>0.62</td>
<td>$&lt; 0.001$</td>
<td>0</td>
<td>-</td>
<td>$&gt; 0.5$</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>(0.44, 0.72)</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>$\log_e$(Bristol Bay sockeye abundance)</td>
<td>1.z</td>
<td>10</td>
<td>7</td>
<td>0.56</td>
<td>$&lt; 0.001$</td>
<td>0</td>
<td>-</td>
<td>$&gt; 0.5$</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>(0.35, 0.58)</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>2.z</td>
<td>5</td>
<td>5</td>
<td>0.47</td>
<td>$&lt; 0.001$</td>
<td>0</td>
<td>-</td>
<td>$&gt; 0.5$</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>(0.39, 0.59)</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Physical or biological variable</td>
<td>Age-class stanza</td>
<td>N</td>
<td>Number of significant positive correlations</td>
<td>Median $r$ for significant positive correlations (range)</td>
<td>$P_{(r)}$</td>
<td>Number of significant negative correlations</td>
<td>Median $r$ for significant negative correlations (range)</td>
<td>$P_{(r)}$</td>
</tr>
<tr>
<td>--------------------------------</td>
<td>-----------------</td>
<td>---</td>
<td>--------------------------------------------</td>
<td>------------------------------------------------</td>
<td>--------</td>
<td>--------------------------------------------</td>
<td>---------------------------------------------------</td>
<td>--------</td>
</tr>
<tr>
<td>PDO</td>
<td>1.z</td>
<td>20</td>
<td>10</td>
<td>0.45 ($0.38, 0.50$)</td>
<td>$&lt;0.001$</td>
<td>0</td>
<td>-</td>
<td>$&gt;0.5$</td>
</tr>
<tr>
<td></td>
<td>2.z</td>
<td>13</td>
<td>6</td>
<td>0.45 ($0.37, 0.53$)</td>
<td>$&lt;0.001$</td>
<td>1</td>
<td>-0.39 ($-0.59, -0.41$)</td>
<td>$&gt;0.2$</td>
</tr>
<tr>
<td>SST</td>
<td>1.z</td>
<td>20</td>
<td>9</td>
<td>0.46 ($0.33, 0.54$)</td>
<td>$&lt;0.001$</td>
<td>3</td>
<td>-0.47 ($-0.59, -0.41$)</td>
<td>0.013</td>
</tr>
<tr>
<td></td>
<td>2.z</td>
<td>13</td>
<td>5</td>
<td>0.49 ($0.37, 0.58$)</td>
<td>$&lt;0.001$</td>
<td>0</td>
<td>-</td>
<td>$&gt;0.5$</td>
</tr>
<tr>
<td>Body size of Bristol Bay age-x.2 recruits</td>
<td>1.z</td>
<td>7</td>
<td>0</td>
<td>-</td>
<td>$&gt;0.5$</td>
<td>3</td>
<td>-0.42 ($-0.29, -0.51$)</td>
<td>$&lt;0.001$</td>
</tr>
<tr>
<td></td>
<td>2.z</td>
<td>5</td>
<td>0</td>
<td>-</td>
<td>$&gt;0.5$</td>
<td>0</td>
<td>-</td>
<td>$&gt;0.5$</td>
</tr>
</tbody>
</table>
Table 1.3 Results of multiple linear regressions of Kalman filter $a_r$ values (Y-intercept) for the sibling models on physical and biological variables for sockeye salmon (*Oncorhynchus nerka*) stocks in British Columbia and Alaska. The fraction of stocks is listed for which each explanatory variable was included in a stepwise add-in procedure ($\alpha \leq 0.05$ for inclusion). SST is the deviation in sea-surface temperature from the winter mean in the Gulf of Alaska. PDO is the Pacific Decadal Oscillation index.

<table>
<thead>
<tr>
<th>Age-class stanza</th>
<th>Fraction of stocks for which each explanatory variable is included in the stepwise procedure for the regression</th>
<th>Mean $r^2_{adj}$ (range)</th>
<th>Mean lag-1 autocorrelation coefficient of residuals (range)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Log$_c$(total sockeye abundance) Log$_c$(Bristol Bay sockeye abundance) SST PDO</td>
<td></td>
<td></td>
</tr>
<tr>
<td>1.z</td>
<td>18/20 1/10 12/20 0/20</td>
<td>0.48 (0.14, 0.64)</td>
<td>0.66 (0.01, 0.87)</td>
</tr>
<tr>
<td>2.z</td>
<td>11/13 0/5 5/13 2/13</td>
<td>0.38 (0.08, 0.67)</td>
<td>0.72 (0.16, 0.95)</td>
</tr>
<tr>
<td>Total cases</td>
<td>29/33 1/15 17/33 2/33</td>
<td>0.45 (0.08, 0.82)</td>
<td>0.69 (0.01, 0.94)</td>
</tr>
</tbody>
</table>
Table 1.4 Summary of correlation coefficients among physical and biological variables.
PDO is the Pacific Decadal Oscillation index. SST is the deviation in sea-surface temperature from the winter mean in the Gulf of Alaska. An asterisk indicates significant correlations at $\alpha = 0.05$.

<table>
<thead>
<tr>
<th></th>
<th>PDO</th>
<th>SST</th>
<th>Log$_e$(total sockeye abundance)</th>
</tr>
</thead>
<tbody>
<tr>
<td>SST</td>
<td>0.16</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Log$_e$(total sockeye abundance)</td>
<td>0.26</td>
<td>0.32</td>
<td>-</td>
</tr>
<tr>
<td>Log$_e$(Bristol Bay sockeye abundance)</td>
<td>0.46*</td>
<td>0.17</td>
<td>0.58*</td>
</tr>
</tbody>
</table>
Figure 1.1  Sibling-age-class relation.
Sibling-age-class relation between the logarithm of the abundance of age-1.2 adult sockeye salmon (*Oncorhynchus nerka*) recruits (fish spending one winter in freshwater and two winters in the ocean) in year $y$, and age-1.3 recruits (fish spending one winter in freshwater and three winters in the ocean) in year $y+1$, for Chignik Lake, Alaska Peninsula for 1918 to 1994 ($r = 0.52$, $P < 0.001$).
Figure 1.2 Distribution of ocean-entry points for the 24 sockeye salmon (*Oncorhynchus nerka*) stocks analyzed, categorized by region. These regions are: Fraser River, British Columbia (B.C.) (denoted with a solid diamond); Central B.C. (solid star); Northern B.C. (open diamonds); Prince William Sound, Alaska (solid circle); Cook Inlet, Alaska (solid triangle); Kodiak, Alaska (open squares); Chignik, Alaska (open circles); Alaska Peninsula (open triangles); and Bristol Bay, Alaska (solid squares).
Figure 1.3  Kalman filter estimates of the Y-intercept for the age-1.2 sibling model (i.e., the \( a_t \) parameter) for two example sockeye salmon (\textit{Oncorhynchus nerka}) stocks, Naknek River, Alaska (a), and Skeena River, British Columbia (b), as a function of brood year. Smoothed at values (solid squares) and filtered at values (open circles) are shown.
Figure 1.4  Annual standardized Y-intercept (i.e., the smoothed $a_t$ parameter) for the age-1.2 sibling model (i.e., fish that spent one winter in fresh water) for Alaska and British Columbia sockeye salmon stocks, as a function of brood year, $t$. Graphs are arranged roughly from northern regions at the top left to southern regions at the bottom right. Each line represents the $a_t$ series for one stock. Note the different time periods covered for each stock, especially those for Chignik and Black Lake stocks in the Chignik area, which are over twice as long as the periods for other stocks.
Figure 1.5 Annual standardized Y-intercept (i.e., the smoothed $a_t$ parameter) for the age-2-z sibling model (i.e., fish that spent two winters in fresh water) for Alaska and British Columbia sockeye salmon ($Oncorhynchus nerka$) stocks, as a function of brood year, $t$.

Graphs are arranged roughly from northern regions at the top left to southern regions at the bottom right. Each line represents the at series for one stock, except for the horizontal lines for Bristol Bay and Kodiak, which applied to three and two stocks, respectively. Note the different periods covered for each stock.
Figure 1.6 Ratios of the mean-squared errors (MSEs) of forecasts from the Kalman filter model to the MSEs of forecasts from the standard sibling model for each sockeye salmon stock.

Black bars represent the 1.2 stanza, open bars represent the 2.2 stanza. Forecasts were produced by a retrospective analysis of 24 stocks across British Columbia and Alaska. Sufficient age-2.2 data were not available for 6 stocks (Pitt River, Long Lake, Skeena River, Copper River, Nushagak, and Nuyakuk). The dashed line at a ratio of 1.0 shows where the MSE of the Kalman filter’s forecasts is equal to that of the standard sibling model.
Figure 1.7  Mean percent bias (%) (see text) of forecasts for the Kalman filter model (black bars) and the standard sibling model (white bars) from a retrospective analysis of age-1.2 recruits for 24 stocks of sockeye salmon (*Oncorhynchus nerka*) across B.C. and Alaska.

*Mean percent bias for Nelson River was off the scale at 598% and 364% for Kalman filter and standard sibling models, respectively.*
Figure 1.8  Mean percent bias (%) of forecasts for the Kalman filter model (black bars) and the standard sibling model (white bars) from a retrospective analysis of age-2.2 recruits for 18 stocks of sockeye salmon (*Oncorhynchus nerka*) across British Columbia and Alaska.

* Mean percent bias for Nelson River was off the scale at 144% for the standard sibling model.
### Figure 1.9

Correlation matrix of smoothed $a_t$ values for the age-1-z sibling model as estimated by the Kalman filter model for sockeye salmon (*Oncorhynchus nerka*) stocks across British Columbia and Alaska. Stocks with constant $a_t$ values were excluded from the analysis. AP on the top row denotes the region of the Alaskan Peninsula; PWS denotes the region of Prince William Sound; BC denotes British Columbia.
Figure 1.10 Correlation matrix of smoothed $a_t$ values for the age-2.2 sibling model as estimated by the Kalman filter model for sockeye salmon (*Oncorhynchus nerka*) stocks across British Columbia and Alaska. Stocks with constant $a_t$ values were excluded from the analysis. AP on the top row denotes the region of the Alaskan Peninsula; CI denotes the region of Cook Inlet; BC denotes British Columbia.
Figure 1.11  Means of correlations between pairs of sockeye salmon (Oncorhynchus nerka) stocks in their time series of smoothed Kalman-filter $a_r$ values (black bars), as well as between their residuals from standard sibling models (white bars), within and among the eight regions. Both age-1.$z$ and age-2.$z$ stanzas are shown.
Preface to Chapter 2

In the previous chapter, I quantified the effects of one type of variability in the biological system (specifically in age-at-maturity of sockeye salmon) on parameter uncertainty in the sibling model and on the resulting accuracy and bias in forecasts of recruitment. Standard sibling models typically assume constant parameters over time, whereas I developed a Kalman filter model that takes into account long-term trends in parameters associated with trends in the distribution of ages-at-maturity. In a retrospective analysis, the Kalman filter model more accurately forecasted adult recruitment than the standard model in 29-39% of stocks analyzed from British Columbia and Alaska, depending on the age-class, and with less bias for 54-94% of stocks. If accounting for this long-term variability results in more-accurate forecasts of recruitment, managers may be better able to assess stock status prior to the fishing season, and hence choose management targets that achieve objectives more exactly than if this variability were ignored. However, any improvements may be reduced by uncertainties in other components of the fishery system. For instance, in Chapter 2, I address an additional source of uncertainty that has not been adequately accounted for and that may reduce managers' ability to achieve objectives, namely, deviations in outcomes of fishing regulations from targets. In this chapter, I quantify that source of uncertainty for sockeye salmon fisheries on the Fraser River, British Columbia, and develop a method to account for it when identifying management targets and evaluating management alternatives.
Chapter 2
Missing the Target: Uncertainties in Achieving Management Goals in Fisheries on Fraser River, British Columbia, Sockeye Salmon (*Oncorhynchus nerka*)

Abstract

In sockeye salmon (*Oncorhynchus nerka*) fisheries, management targets are rarely achieved exactly, thereby creating uncertainties about outcomes from implementing fishing regulations. Although this type of uncertainty may be large, it is seldom incorporated into simulation models that evaluate management options. One objective of this study was to quantify the deviations that occur between realized and target mortality rates (i.e., the target fraction of adult recruits that die each year during return migration, mostly due to harvesting) in fisheries for sockeye salmon from the Fraser River, British Columbia. We found that for some sockeye stocks, realized mortality rates were higher than targets when recruitment was low (resulting in conservation concerns), and lower than targets when recruitment was high (resulting in foregone catch). Scientists and managers can at least partially account for effects of such deviations between realized and target mortality rates (outcome uncertainties) by choosing target harvest rules that reflect typical patterns in those deviations. We derived a method for incorporating those patterns into analyses of management options.

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Introduction

Uncertainties are pervasive in fisheries, and if ignored by managers, they may result in foregone yield or create conservation concerns. One type of uncertainty that has been studied relatively little is the frequently observed difference between a management agency's desired target, such as a harvest rate or abundance of spawners, and the outcome that actually occurs. One source of this difference is the imperfect control that managers have over outcomes after issuing fishing regulations, i.e., "implementation uncertainty" or "implementation error" (Rosenberg and Brault 1993; Kell et al. 1999). That type of uncertainty occurs in part because managers are unable to predict exactly the behaviour of harvesters in response to regulations, and therefore usually cannot take this behaviour into account when setting regulations. Other factors also affect whether targets are met. Examples include temporal variability in catchability of fish (Cass et al. 2003) and numbers of fishing vessels, as well as errors in estimating fish abundance (Rosenberg and Restrepo 1994). These uncertainties about outcomes from particular fishing regulations can be large, and they occur in a variety of fisheries. For example, the annual ratio of actual spawner abundance to the target varied between 0.4 and 2.4 for eight sockeye salmon (Oncorhynchus nerka) stocks in Bristol Bay, Alaska between 1962 and 1982 (Bocking and Peterman 1988). Such variations are economically important. Rice and Richards (1996) suggested that reductions in implementation uncertainty in British Columbia's (B.C.) rockfish fishery between 1993 and 1994 were partly responsible for the 17% increase in commercial value over that period.

Despite this importance of deviations between targets and outcomes, stock assessments and simulation studies of fisheries on Pacific salmon (Oncorhynchus spp.)
and other species have usually implicitly assumed that management targets are met and that regulations are implemented as planned (McAllister et al. 1999). In cases where deviations from targets have been accounted for, most have either assumed that those differences are deterministic (Kell et al. 1999), or if stochastic, that they are unbiased with respect to the target (e.g., Johnston et al. 2000). Furthermore, methods to account for deviations from targets have usually been based more on expert opinion than on data (Peterman et al. 2000; Cass et al. 2003). Some Pacific salmon studies have quantified a few of the many potential sources of deviations between realized mortality rates and targets (e.g., Link and Peterman 1998), which, if only considered on their own, underestimate the total variance in realized mortality rates. Thus, when these and similar uncertainties are ignored or incorrectly represented, modelers may be providing managers with inappropriate advice about the effectiveness of certain management actions.

To provide more appropriate advice, we need a clearer understanding of the differences between management targets and observed outcomes, as well as the causes of those differences in order to reduce them. In various Pacific salmon fisheries in North America, management goals are typically stated in terms of either target spawner abundances (target escapements) or target fractional mortality rates (i.e., the fraction of total adult returns projected to die during return migration, mostly due to harvesting). Although the differences between these targets and the actual (i.e., realized) spawner abundances or mortality rates have been referred to by most authors as implementation uncertainties, these differences are not just affected by lack of control over harvesting. For example, they are also influenced by variable mortality during upstream migration (Cooke et al. 2004) and errors in pre-season abundance estimates of the potentially
fishable stock (Rosenberg and Restrepo 1994). Thus, the term implementation uncertainties or errors may have been used too broadly in the past.

In this study, we empirically estimated the properties of a broader variable, the difference between realized and target mortality rates (outcome uncertainty), and examined several potential sources of those differences, only one of which is implementation uncertainty. We focused on fisheries for sockeye salmon from the Fraser River, B.C., Canada. Recently, managers there have chosen annual target harvest rates based on a combination of forecasts of recruitment (mature adults that return to the coast toward natal spawning areas), estimated productivity (potential rate of change) of sockeye populations, pre-specified guidelines regarding harvest rate plus natural mortality rate, and escapement targets (Fraser River Sockeye Spawning Initiative 2005). For the rest of this paper, we use the term mortality rates instead of harvest rates (i.e., catch/recruitment) because our estimates of realized, or actual, recruitment reflect losses of fish due to factors other than harvesting, such as en-route mortality from high water temperatures or parasites (Williams 2005). These losses are particularly relevant for some stocks of Fraser River sockeye salmon.

For Fraser River sockeye, there are at least six potential causes for differences between realized and target mortality rates, the first two of which pertain specifically to implementation uncertainty: (1) the degree of compliance with regulations by harvesters, (2) deviations in catchability coefficients from those assumed in assessments (Cass et al. 2003) due to variation in migratory behaviour of fish, environmental conditions, and the efficiency of fishing gear, (3) deviations from initial target mortality rates caused by managers making deliberate trade-offs during the fishing season among management
objectives (e.g., among biological escapement, social, and economic objectives) (Wood et al. 1998; Cooke et al. 2004), (4) deviations from the expected natural mortality that occur between the time of harvest and when spawning escapement is estimated (i.e., en-route mortality) (Cooke et al. 2004), (5) estimation errors in catch and spawner abundances (Pacific Salmon Commission 2002), and (6) incorrect forecasts of abundances that affect calculations of target mortality rates (Bocking and Peterman 1988; Cass et al. 2003).

A closer examination of the latter (forecasting errors) is warranted, given that in the Fraser River sockeye fishery, it is the source of differences in realized mortality rates from targets that scientists can, in theory, reduce through improved forecasting methods. Forecasts of abundance of total adult sockeye recruits are typically made prior to the fishing season based on various quantitative analyses (Cass 2002), and forecasts are updated during the season using observations from test, commercial, and First Nations fisheries, as well as other sources (Pacific Salmon Commission 2001). As forecasts are updated during the season, management targets are adjusted accordingly.

For sockeye salmon, the direction and magnitude of deviations in realized mortality rates from targets are partly related to both actual recruitment and the direction and magnitude of forecasting errors in recruitment (Bocking and Peterman 1988). For instance, within-season updates of estimated recruitment often occur too late to allow managers to close areas to fishing to correct entirely for forecasts that overestimate abundance, resulting in overexploitation at low recruitment. In contrast, when actual recruitment is higher than the forecast, within-season updates tend to be too late to permit managers to allow enough fishing in various areas to prevent under-exploitation.
Furthermore, in this latter case, differences between realized and target mortality rates may be further increased at very large recruitment due to limited fishing and processing capacity in remote areas such as Bristol Bay, Alaska (Bocking and Peterman 1988).

If there is a repeatable pattern to the magnitude and direction of deviations in realized mortality rates from targets, then capturing that pattern in simulations that evaluate management options can help managers meet their objectives. Specifically, when simulation models are used to quantitatively evaluate management options, a more accurate representation of those deviations will result in more appropriate estimates of the effects of management options. Ignoring those outcome uncertainties may lead to decisions that do not achieve management objectives.

Research objectives

Our research objectives were therefore to (1) quantify the properties (magnitude and nature) of deviations between realized mortality rates and targets in four aggregates of stocks of Fraser River sockeye salmon, (2) examine the contribution to these deviations by forecasting errors, environmental factors, and specific management situations, and (3) derive a method to permit modelers to incorporate those observed properties into simulations that evaluate management options.

Methods

Instead of comparing annual targets to annual realized outcomes for Fraser River sockeye salmon fisheries, we characterized the differences between target and realized mortality rates by comparing target harvest rules (i.e., state-dependent and time-independent relations that describe changes in target mortality rates or escapement goals).
with realized harvest functions that describe the actual mortality rate or escapement that occurred by the end of the fishing season. Specifically, we compared two relations (Fig. 2.1). The first was desired fractional mortality rate as a function of forecasted recruitment (referred to as the *target harvest rule*). Such state-dependent harvest rules are now becoming commonplace in fisheries (Butterworth and Punt 1999). They allow for feedback between target mortality rates and the estimated fishable stock. The second relation was realized fractional mortality rate as a function of forecasted recruitment (the *realized harvest function*) (Fig. 2.1). By comparing these two relations, we examined patterns of bias in the difference between realized mortality rates and targets that are expected to emerge over the long term, independent of year-to-year variability.

Note that although managers of many sockeye salmon fisheries often use constant escapement goals instead of target harvest rules, these two concepts are closely linked. Below a threshold abundance of adult recruits (the target escapement or the X-intercept for the target harvest rule in Fig. 2.1) the target harvest is zero, and above this threshold abundance, all "surplus" recruits are harvested, which produces nonlinearly increasing harvest rates as recruitment increases.

**Data**

We obtained data from various management-agency sources (Table 2.1) on abundance of adult sockeye salmon recruits and spawners for 1986 through 2003 for four stock complexes from the Fraser River, B.C., categorized by the timing of their upstream adult migration. Below we describe how data were generated by management agencies.
Estimates of realized adult recruitment were provided by the Canada Department of Fisheries and Oceans (DFO) and the Pacific Salmon Commission (PSC) (Appendix). Those estimates included the sum of estimated spawning escapement and catch from marine and in-river commercial and test fisheries, Fraser River First Nations harvests, and catch from recreational fisheries, as well as natural en-route mortality, fishing-induced mortality that was not included in catch data, and estimates of unreported catch. Pre-season forecasts of recruits from DFO, which were based on abundances of various age groups and environmental indices (e.g., Cass 1998, 2002), were updated during the fishing season. We did not use pre-season forecasts but instead only used the final annual within-season forecasts of recruits that could be applied by managers to update target mortality rates during the fishing season. This was because those forecasts tend to be more accurate than pre-season forecasts and therefore result in realized mortality rates that are closer to targets.

Within-season target mortality rates (and corresponding target escapements) for Fraser River sockeye salmon (Appendix) were chosen annually by DFO from predetermined decision rules according to a specified percentile (e.g., 50th or 75th) of the probability distribution for forecasts of recruitment. In recent years, additional decision rules were used to keep target mortality rates below a maximum threshold (e.g., no greater than 15% mortality rate in some years to protect certain stocks that were a conservation concern) (Pacific Salmon Commission 2001).
Statistical analysis

Model structure

We used a three-parameter model (equation 1) to describe the relation between forecasts of adult recruitment in year \( t \), \( R'_t \), and either the target fractional mortality rate, \( D'_t \), (i.e., the fraction expected to die during return migration), or \( D_t \) (the fraction that actually died). For the latter, the realized mortality function is:

\[
D_t = a(1 - e^{b(c-R'_t)}) \quad \text{for } R'_t > c , \text{ and }
\]

\[
D_t = 0 \quad \text{for } R'_t \leq c ,
\]

where \( a \) is the maximum mortality rate at high recruitment, \( b \) is a shape parameter, and \( c \) is the protection level (minimum escapement target) below which mortality rate is zero (e.g., Fig. 2.1). We chose this model form because its parameters were directly relevant to managers, and it adequately described the data. The model structure of this realized harvest function was identical to that of the target harvest rule, except the latter calculated target mortality rate, \( D'_t \), again as a function of forecasts of recruitment, \( R'_t \), but using the parameters \( a' \), \( b' \), and \( c' \), instead of \( a, b, \) and \( c \). These models reflect the average target harvest rule or realized harvest function over all years as a function of forecasts of recruitment.
Parameter estimation

The relation among the fraction of fish dying, escapement, and recruitment (for example, for the realized mortality rate, $D_i$) is:

$$D_i = \left( \frac{R_i - E_i}{R_i} \right),$$

(2)

where $R_i$ is realized recruitment and $E_i$ is realized spawning escapement, both based on post-season data provided by management agencies (Appendix). To estimate the parameters of the target harvest rule and realized harvest function for each sockeye run-timing group, we rearranged equation 2 to solve for escapement and added an error term:

$$E_i = [R_i(1 - D_i)] \cdot e^{\nu},$$

(3)

where $\nu$ has a mean of zero and variance $\sigma^2$. We added this multiplicative lognormal error term ($e^{\nu}$) for parameter estimation purposes to reflect the typically observed lognormal distribution for recruitment and escapement of Pacific salmon (Hilborn and Walters 1992). Substituting equation 1 into equation 3 allowed us to estimate the parameters of the realized harvest function, $a$, $b$, and $c$, from empirical data (Appendix) on $E_i$, $R_o$, and $R'_o$, with the latter entering through equation 1. Although this formulation of the model did not explicitly constrain mortality rates to be between zero and one, the best-fit parameters always gave predictions of $D_i$ from equation 1 that were within that
range. To estimate the parameters, we used a maximum likelihood procedure and a robust method, a Cauchy M-estimator, because the latter down-weights the influence of outliers and, as opposed to other robust methods, it has a continuous derivative, thereby producing parameter estimates that are more stable (Huber 1981). We followed an analogous procedure to estimate parameters of the function describing target mortality rates.

**Other analyses**

We then investigated three potential sources of variability in deviations between realized mortality rates and targets predicted from the harvest rule: forecasting errors, variability in environmentally driven en-route mortality, and specific management situations. These sources of outcome uncertainty were not mutually exclusive, e.g., closures to fishing might occur when unfavourable environmental conditions exist.

We examined the extent to which differences between realized mortality rates (equation 2) and target mortality rates (determined by the target harvest rule -- equation 1 but with \(D', a', b', \text{ and } c'\)) were due to variability in forecasting errors for Fraser River sockeye fisheries. We fit a linear relation between those differences and forecasting errors (as defined by the ratio of within-season forecasts of recruitment to actual realized recruitment).

Realized mortality rates of adult recruits may also deviate from targets due in part to interannual variability in environmentally driven "en-route mortality". Specifically, warm temperatures and high water discharge in the Fraser River increase (a) energy expenditure of adult sockeye salmon during up-river migration, (b) the duration of this
migration, and (c) the prevalence and severity of parasites and disease, which can result in dramatically increased mortality (Wagner et al. 2005; Williams 2005). In years when water temperature and discharge exceed critical thresholds for salmon survival, managers typically reduce fishing pressure to attempt to achieve management targets (Pacific Salmon Commission 2005; Ian Guthrie pers. comm.). To reflect this situation, we examined two different thresholds in annual environmental conditions (Pacific Salmon Commission --Ian Guthrie, pers. comm.; and Fisheries and Oceans Canada -- David Patterson, pers. comm., Cooperative Resource Management Institute, Simon Fraser University, 8888 University Drive, Burnaby, British Columbia, V5A 1S6, Canada, and J. Stevenson Macdonald, Center for Aquaculture and Environmental Research, 4160 Marine Drive, West Vancouver, British Columbia, V7V 1N6, Canada). The first threshold for increased sockeye mortality occurred in years when two conditions were met: river temperatures were greater than 18.0 °C at Hell's Gate, B.C., when averaged over the 31-day period centered around the peak migration period at Hell's Gate, and mean daily temperature exceeded that value for at least 18 days during this period. The second environmental threshold occurred in years when mean water discharge at Hell's Gate was greater than 6400 m³·s⁻¹ averaged over the same period.

Certain management situations may also cause target and realized mortality rates to differ. Managers of Fraser River sockeye salmon specify targets for the four run-timing groups individually due their differences in productivity. However, in practice, it is difficult to separately manage these stocks because fishing pressure on one run-timing group affects others due to spatial and temporal overlap of some stocks. To investigate
this source of variability, we determined whether deviations between realized and target mortality rates were significantly correlated among run-timing groups.

**Conversion factors**

If the target harvest rule and realized harvest function for a given stock group differ in a systematic manner (i.e., if differences between realized and target mortality rates have a mean other than zero), then representing that bias in stock assessment models of salmon fisheries may improve predictions of outcomes from applying various harvest strategies. To this end, we estimated stock-specific scalars, or conversion factors ($d_0, d_1,$ and $d_2$), that transformed the maximum likelihood parameter values for the target harvest rules ($a', b', c'$) into those of the realized harvest functions ($a, b, c$) (equation 1). These conversion factors for any one stock group were estimated by first substituting parameters of the target harvest rule ($a', b',$ and $c'$) and their unknown respective scalars into the realized harvest function (i.e., substituting $a = d_0 \cdot a', b = d_1 \cdot b', c = d_2 \cdot c'$ into equation 1 and then substituting it into equation 3). To estimate the best-fit values of $d_0, d_1,$ and $d_2$, we then used the previously estimated values for $a', b'$, and $c'$ along with the same maximum likelihood estimation procedure as noted above that assumed log-normally distributed random errors for equation 3.

**Results**

The nature of the deviations between realized and target mortality rates differed among run-timing groups. Specifically, for the Late Summer run, the fitted realized harvest function showed higher fractional mortality rates than the fitted target harvest rule for all historically observed levels of recruitment (Fig. 2.2d). In particular, for that run-
timing group, the 95% confidence limits for $a'$ (maximum mortality rate at high recruitment for the target harvest rule) did not include the best estimate for the $a$ parameter from the realized function (Table 2.2). Because our purpose was to describe patterns in average deviations for use in future modelling and not to test statistical hypotheses about model fits, we state our comparisons of parameters in this way to concisely describe the results. The presence of overlap between 95% confidence intervals of two parameters (or between the interval of one parameter and the best-fit value of the other) was not critical for our analyses. In the year of the largest observed actual recruitment from 1986 through 2003 (12.4 million fish), the realized harvest function predicted a realized catch of 9.4 million fish per year instead of 7.7 million fish predicted from the target harvest rule. These numbers were based on a realized mortality rate of 76% and the target mortality rate of 62%.

For the Early Stuart run, at low recruitment, realized mortality rates also tended to be higher than target mortality rates, but the reverse was true at high recruitment (Fig. 2.2a). For instance, at $R' = 0.25$ million fish, the realized mortality rate predicted from the function was higher than the target predicted from the target harvest rule (0.63 compared to 0.41), but at $R' = 1.5$ million fish, the realized mortality rate predicted from the function was lower than the target (0.63 compared to 0.70). For this Early Stuart run-timing group, the 95% confidence limits for $b'$, which reflected the rate at which target mortality rates increased with forecasts of recruitment, did not include the best estimate for $b$ from the realized harvest function (Table 2.2). For the Early Summer and Summer runs, the differences between the target harvest rules and realized harvest functions were
small and the 95% confidence limits for all parameters of the target harvest rules included the best estimates of parameters of the realized harvest functions.

In sensitivity analyses, these observed patterns in deviations between realized and target mortality rates were virtually unaffected by two key uncertainties, the error structure in escapement (normally distributed versus lognormally distributed in equation 3) and cyclic patterns in recruitment. We considered the latter uncertainty because many Fraser River sockeye salmon populations follow four-year cycles of abundance in which the "dominant cycle line" and "sub-dominant cycle line" have higher abundances than the two "off-cycle lines", and these patterns of abundances may affect the accuracy with which managers achieve target escapements (Cass et al. 2003). However, the direction of bias in deviations between target harvest rules and realized harvest functions did not change when we divided the recruitment and escapement data into dominant/sub-dominant and off-cycle lines.

These results illustrate that outcome uncertainties can be substantial, and for at least one run-timing group (Early Stuart), those differences between realized and target mortality rates show bias that is a nonlinear function of recruitment. In addition, annual data points indicate that there was substantial variability around such underlying functions (Fig. 2.2). For example, when averaged across run-timing groups, the coefficient of variation in residuals of realized mortality rates from the mortality rates predicted from equation 1 was 0.35 (ranging from 0.24 to 0.59). Furthermore, the magnitude and direction of deviations in realized mortality rates from targets (identified from the target harvest rule) appears to have changed over time for all four run-timing groups (Fig. 2.3).
Effects of forecasting errors

Stocks differed in the degree of association between forecasting errors and deviations between realized mortality rates and targets, the latter identified from target harvest rules. We fit a linear relation between these deviations in mortality rates and forecasting errors. For the Early Stuart run, this linear relation was marginally not significant ($F= 4.21, P = 0.057, \text{df} = 17$), and, based on adjusted $r^2$, 16% of the variance in deviations in mortality rates could be explained by forecasting errors. Residuals from this relation were normally distributed. For the Early Summer, Summer, and Late Summer runs, less than 1% of the variance in differences between realized and target mortality rates could be explained by this source of uncertainty, and these relations were not statistically significant ($P = 0.96, 0.36, \text{and} 0.87$, respectively). Therefore, factors other than uncertainties in forecasts of recruitment likely accounted for most of the variance in deviations between realized and target mortality rates for these run-timing groups. We explored three possible factors, one related to environmentally driven en-route mortality and two related to specific management situations.

Effects of environmentally driven en-route mortality

For Early Stuart, Early Summer, and Summer runs, in years when water flows and river temperatures exceeded critical thresholds (Fig. 2.4, right sides of the six panels), realized mortality rates tended to be greater than those from the target harvest rules (positive residuals). In fact, for those stocks and years for which water flow exceeded the critical threshold, realized mortality rates were always greater than within-season targets. For those stocks and years for which river temperature exceeded the critical threshold, there also tended to be more positive deviations in mortality rates than expected if those
deviations were independent of temperature, although the pattern was not statistically significant (binomial test, $P = 0.22$) (Zar 1999). No such anomalous environmental conditions occurred for the Late Summer run during the period analyzed; hence, this stock is omitted from Figure 2.4.

**Effects of specific management situations**

For the Early Summer run, differences in realized mortality rates from targets were positively correlated with those of the Summer run ($r = 0.66$, $P < 0.01$) because timings of up-river migration (when the fish are vulnerable to the main fishery) partially overlapped for these two runs (Pacific Salmon Commission 2001). Fishing pressure on the Early Summer run to some extent affected realized mortality rates on the Summer run, and vice versa.

Furthermore, in the seven years when Late Summer sockeye migrated earlier than normal, it partly coincided with the Summer run. In those years, realized mortality rates for the Summer run were lower than predicted from the target harvest rule for that stock group (hollow circles, Figs. 2.4e and 2.4f). This was because mortality rates were affected by deliberate management decisions to reduce fishing pressure on the co-migrating Late Summer run, which faced conservation concerns. In contrast, for the Late Summer run, there was no consistent sign of deviations in mortality rates in the years when this run migrated earlier than expected (not shown).

**Conversion factors**

Table 2.3 lists the factors that convert target harvest rules to realized harvest functions. For Early Summer, Summer, and Late Summer runs, the differences between
the target harvest rule and realized function were small, so conversion factors for most parameters for those stocks were close to one. The Early Stuart run was the exception.

Discussion

For some sockeye salmon fisheries, actual fractional mortality rates tend to deviate from targets in ways that may hinder achievement of biological conservation objectives and harvesting objectives. For example, for Early Stuart and Late Summer run-timing groups, realized mortality rates generally tended to exceed target mortality rates predicted from target harvest rules at low recruitment (though at all recruitments for the Late Summer run). Where such a bias is not accounted for in pre-season stock assessments and evaluations of management options, salmon populations are being put at risk in terms of conservation concerns more often than would be the case if this bias were taken into account. Even though in some cases harvesting occurs at higher rates than initially targeted because of deliberate trade-offs among management objectives in multi-stock fisheries, this difference between target and realized mortalities should nevertheless be considered when evaluating management options.

Attainment of economic and social objectives may also be affected by this bias in differences between realized mortality rates and targets. For instance, for the Early Stuart run, the realized mortality-rate function tends to fall below the target mortality rate function at high recruitment. In that situation, when recruitment was greater than 1.4 million fish, for example, the realized mortality rate predicted from the realized harvest function was 89% of the target mortality rate. This bias would result in foregone catch when recruitment was high, leading to reduced economic and social benefits associated with harvesting. In future evaluations of management options for Fraser River sockeye
salmon stocks, scientists and managers can account for both positive and negative biases in realized minus target mortality rates by using models with conversion factors like those identified here.

Another salient feature of our results is the large interannual variation in the magnitude of outcome uncertainties. These large variations mean that the best-fit conversion factors would need to be combined with random error terms to simulate this observed stochasticity in realized mortality rates when evaluating management options. Such stochastic "noise" may be so large as to mask in the short term the differences between forecasted harvest rules and realized harvest functions, but in the long term, the patterns in deviations between realized mortality rates and targets will emerge. For instance, for the Late Summer run, realized mortality rates will on average be greater than target mortality rates, though this will not necessarily be the case in any given year.

Although differences between realized mortality rates and targets can be partially explained by forecasting errors, environmentally driven en-route mortality, and specific management situations, these apply only to certain stocks, and much of the total variance remains unexplained. Forecasting errors could only explain a small portion of the variance in deviations between realized and target mortality rates, and environmental conditions could only explain those deviations in years when conditions surpassed critical thresholds. Therefore, other sources of outcome uncertainty are likely important.

For instance, measurement errors in spawner abundances may contribute to variability in estimated in-river mortality rates. The coefficient of variation (CV) in estimates of spawner abundances for nine sockeye salmon stocks in the Fraser River, as derived from mark-recapture studies, ranged from 0.02 to 0.19 from 2001 through 2004...
These measurement errors were lower than the coefficient of variation that we calculated for deviations between realized spawning escapements and targets identified from target harvest rules (average $CV=0.65$, ranging from 0.47 to 0.82). Therefore, it is likely that estimation error is only a minor contributor to the observed differences between target and realized mortality rates.

Implementation uncertainty is another possibility for those differences. For example, non-compliance by harvesters with regulations may explain part of the observed high realized mortality rates at low recruitment compared to the target. However, we have no data to further analyze this source of implementation uncertainty. Similarly, deviations in catchability coefficients may also explain part of the bias and stochasticity in observed mortality rates because migratory routes of adults vary unpredictably among years (Pacific Salmon Commission 2001).

One limitation to our results is that our conversion factors represent the average deviation over 18 years between realized mortality rates and targets that were identified from harvest rules. However, the magnitude and direction of these differences appear to have changed several times over this period, possibly due to a combination of factors. These include changes in forecasting methods, management objectives, characteristics of the fishing fleet and allocation of harvest among various groups, methods used to estimate abundances of spawners and the catch, environmental factors, en-route mortality, and migratory behaviour of recruits. Conversion factors such as ours therefore only reflect long-term average patterns in differences between realized and target mortality rates.
Some previous studies of sockeye salmon examined biases in differences between realized harvest rates (or spawning escapements) and targets. However, they generally have been based more on either expert opinion or qualitative perusal of data than quantitative analyses of empirical data (e.g., Peterman et al. 2000). Bocking and Peterman (1988) identified differences in annual realized harvest rates from annual target harvest rates (or annual realized escapements from annual target escapements), rather than deviations in actual outcomes from underlying target harvest rules that are time-independent. In contrast to Bocking and Peterman's (1988) approach, Wood et al. (1998), who examined sockeye salmon fisheries on the Skeena River, B.C., attributed the deviations between target harvest rules and actual harvest rates from 1983 through 1997 to trade-offs among objectives by managers in a mixed-stock fishery, although these deviations were not described quantitatively. Furthermore, empirical studies of Pacific salmon fisheries have usually ignored the effect of abundance of recruits on the direction and magnitude of deviations between realized mortality rates and targets (Wood et al. 1998).

Our measure of variability in the difference between realized and target mortality rates was lower than that estimated from another previous study (for sockeye salmon in Bristol Bay, Alaska, Eggers 1992), but higher than others (e.g., for sockeye salmon in Fraser River, B.C., Cass et al. 2003). The difference in the magnitude of Eggers' (1992) estimates and our values for Fraser River run-timing groups may be due to reduction in forecasting error through managers using within-season updates of recruitment forecasts in the Fraser River cases that were not used in Eggers (1992) for the Bristol Bay fishery. The estimate of variability used by Cass et al. (2003) was based on expert judgment and
included only those sources of uncertainty related to variability in catchability of fish, whereas ours was based on empirical data and reflected various sources of uncertainty, including forecasting errors and errors in estimates of spawner abundance.

Management recommendations

Scientists and managers can account for outcome uncertainties, or differences between realized mortality rates and targets, in several ways. First, they can identify appropriate target rules that reflect those differences by building stochastic simulations models. The Fraser River Salmon Spawning Initiative model is being developed with this mind (Cass et al. 2003). Managers can also reduce those differences by identifying the sources and applying corrective actions. For example, more accurate within-season forecasts of recruitment may bring realized mortality rates closer to targets. For the Fraser River sockeye salmon, managers have attempted this by predicting en-route mortality with newly developed management-adjusted (MA) forecasts of spawning escapement. Those MA forecasts account for natural, as well as other, sources of en-route mortality, such as in-river First Nations fisheries, estimation errors in in-river abundance and at the spawning grounds, and stock identification errors. By doing so, managers aim to reduce deviations between realized mortality rates and targets (Pacific Salmon Commission 2005). For example, for the 2004 forecast of spawning escapement for Early Stuart sockeye that included this adjustment was 24% lower than the forecast that ignored it (Ian Guthrie pers. comm.). MA forecasts have been implemented in the Fraser River only recently (starting in 2002), and therefore it is too soon to determine their effect on deviations between realized mortality rates and targets. In addition, more effective enforcement of regulations and monitoring of compliance may further reduce
implementation uncertainty by deterring harvesters from violating regulations. An
example could be to use genetic techniques to identify fish harvested illegally from
stocks in areas closed to fishing (Withler et al. 2004). Collectively, these steps should
help achieve management objectives more frequently than is currently the case. At the
very least, accounting for differences between realized and target mortality rates in
simulations of management options should be of value to managers of most fish species.

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References

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(Oncorhynchus nerka): comparison of methods and economic considerations.

Butterworth, D. S., and Punt, A. E. 1999. Experiences in the evaluation and


### Table 2.1 Sources of sockeye salmon data for the Fraser River, British Columbia that we analyzed for implementation uncertainty.

"Return years for data" is the duration of the time series. Target spawning escapement and forecasts of recruitment are the last within-season values that are used to affect final management decisions. Escapement and recruitment data refer to adults only (i.e., jacks\(^a\) were excluded), unless stated otherwise.

<table>
<thead>
<tr>
<th>Type of data</th>
<th>Return years for data</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>Within-season target spawning escapement</td>
<td>1986-2003</td>
<td>Ian Guthrie(^b)</td>
</tr>
<tr>
<td>Within-season forecasts of recruitment</td>
<td>1986-2003</td>
<td>Ian Guthrie</td>
</tr>
<tr>
<td>Realized spawning escapement</td>
<td>1986-2003</td>
<td>Jeff Grout(^c)</td>
</tr>
<tr>
<td>Realized recruitment including jacks</td>
<td>1996-2003</td>
<td>Ian Guthrie</td>
</tr>
<tr>
<td>Total number of jacks in spawning escapement</td>
<td>1996-2003</td>
<td>Reports of the Fraser River Panel to the Pacific Salmon Commission 1996-2003(^e)</td>
</tr>
</tbody>
</table>

\(^a\)Jacks are males that return to spawn after only one year in the ocean (usually as 3-year-olds).

\(^b\)Personal communication. Pacific Salmon Commission, 600-1155 Robson Street, Vancouver, British Columbia, V6E 1B5, Canada.

\(^c\)Personal communication. Fisheries and Oceans Canada, 100 Annacis Parkway, Unit 3, Delta, British Columbia, V3M 5P8, Canada.


Table 2.2 Parameter values for target harvest rules \((a', b', \text{ and } c')\) and realized harvest functions \((a, b, \text{ and } c)\) for sockeye salmon fisheries in four run-timing groups of the Fraser River, British Columbia. Values in parentheses are the lower and upper 95% confidence limits. For the Early Summer run, the \(c\) parameter was constrained to equal to 1 fish to permit model convergence; therefore, the confidence limits for this parameter were not calculated.

<table>
<thead>
<tr>
<th>Run-timing group</th>
<th>Parameter</th>
<th>(a') (target or (a) (realized))</th>
<th>(b') (target or (b) (realized))</th>
<th>(c') (target or (c) (realized))</th>
</tr>
</thead>
<tbody>
<tr>
<td>Early Stuart</td>
<td>Target harvest rule</td>
<td>0.70 (0.59, 0.82)</td>
<td>3.75x10^{-6} (1.74x10^{-6}, 7.48x10^{-6})</td>
<td>19 486 (1, 82 370)</td>
</tr>
<tr>
<td></td>
<td>Realized harvest</td>
<td>0.63 (0.45, 0.94)</td>
<td>5.52x10^{-3} (2.06x10^{-3}, 4.78x10^{-3})</td>
<td>1.83 (6.7x10^{-3}, 294)</td>
</tr>
<tr>
<td>Early Summer</td>
<td>Target harvest rule</td>
<td>0.75 (0.68, 1.00)</td>
<td>3.48x10^{-6} (1.24x10^{-6}, 9.50x10^{-6})</td>
<td>1.01 (1.00, 214)</td>
</tr>
<tr>
<td></td>
<td>Realized harvest</td>
<td>0.68 (0.56, 0.84)</td>
<td>7.69x10^{-6} (5.23x10^{-7}, 3.92x10^{-5})</td>
<td>1</td>
</tr>
<tr>
<td>Summer</td>
<td>Target harvest rule</td>
<td>0.74 (0.66, 0.84)</td>
<td>1.01x10^{-6} (3.85x10^{-7}, 3.30x10^{-6})</td>
<td>1.17 (1.00, 252)</td>
</tr>
<tr>
<td></td>
<td>Realized harvest</td>
<td>0.75 (0.66, 0.82)</td>
<td>6.93x10^{-7} (2.69x10^{-7}, 2.37x10^{-6})</td>
<td>1.44 (1.06, 404)</td>
</tr>
<tr>
<td>Run-timing group</td>
<td>Parameter</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>------------------</td>
<td>-----------</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>( a' ) (target) or ( a ) (realized)</td>
<td>( b' ) (target) or ( b ) (realized)</td>
<td>( c' ) (target) or ( c ) (realized)</td>
<td></td>
</tr>
<tr>
<td>Late Summer</td>
<td>0.62 (0.52, 0.72)</td>
<td>4.54 \times 10^{-5} (6.57 \times 10^{-6}, 8.02 \times 10^{-3})</td>
<td>1808 (259, 368,952)</td>
<td></td>
</tr>
<tr>
<td>Realized harvest</td>
<td>0.76 (0.67, 0.84)</td>
<td>4.55 \times 10^{-5} (2.06 \times 10^{-9}, 1.21 \times 10^{-7})</td>
<td>1808 (259, 368,952)</td>
<td></td>
</tr>
</tbody>
</table>

function
Table 2.3 Conversion factors.
Conversion factors ($d_0$, $d_1$, $d_2$) that can be used as multiplicative scalars to transform parameters of the target harvest rules ($a'$, $b'$, and $c'$) into parameters of the realized harvest functions ($a$, $b$, and $c$ in equation 1) for four run-timing groups of sockeye salmon from the Fraser River, British Columbia.

<table>
<thead>
<tr>
<th>Run-timing group</th>
<th>$d_0$</th>
<th>$d_1$</th>
<th>$d_2$</th>
</tr>
</thead>
<tbody>
<tr>
<td>Early Stuart</td>
<td>0.89</td>
<td>14.73</td>
<td>$4.75 \times 10^{-5}$</td>
</tr>
<tr>
<td>Early Summer</td>
<td>0.91</td>
<td>2.21</td>
<td>1.00</td>
</tr>
<tr>
<td>Summer</td>
<td>1.01</td>
<td>0.69</td>
<td>0.082</td>
</tr>
<tr>
<td>Late Summer</td>
<td>1.22</td>
<td>1.00</td>
<td>1.33</td>
</tr>
</tbody>
</table>
Figure 2.1 Hypothetical plot of a target harvest rule for sockeye salmon showing the desired fractional mortality rate (fraction dying) as a function of the forecast of recruitment, and the associated realized harvest function showing the fractional mortality rate actually observed as a function of the forecast of recruitment. The parameter $a'$ is the maximum mortality rate at high recruitment for the target harvest rate, and $c'$ is the "protection level" or abundance of recruits below which target mortality is intended to be zero (i.e., the target escapement). Parameters $a$ and $c$ without primes are for the actual, or realized, harvest function.
Figure 2.2  Data and best-fit curves for target harvest rules (i.e., target fractional mortality rate as a function of the forecast of recruitment, solid lines and solid circles), and realized harvest functions (i.e., realized fractional mortality rate as a function of that same forecast of recruitment, dashed lines and open circles) for four run-timing groups of sockeye salmon from the Fraser River, British Columbia for return years 1986 through 2003.
Figure 2.3  Time series of differences between annual realized fractional mortality rates and the targets that were identified from target harvest rules in those years by run-timing group of sockeye salmon from the Fraser River, British Columbia. Early Stuart (solid line), Early Summer (solid line with data points), Summer (short-dashed line), Late Summer (long-dashed line).
Figure 2.4  Relation between environmental variables (mean water flow and mean river temperature at Hell's gate for the 31-day period surrounding peak migration in year t) and the difference between the realized fractional mortality rate, $D_t$, and the target fractional mortality rate, $D_t^*$, that was predicted from target harvest rules, for three run-timing groups of sockeye salmon in the Fraser River, British Columbia.

The horizontal solid lines reflect residuals of zero. The dashed vertical lines represent critical thresholds in water flow ($6400 \text{ m}^3\text{s}^{-1}$) and river temperature ($18^\circ\text{C}$) for increased natural mortality, as specified by the Pacific Salmon Commission. Hollow circles in panels (e) and (f) represent years when the timing of the Late Summer run was early (prior to September 15) and coincided with that of the Summer run. (See following page.)
Early Stuart

(a) Difference in proportional mortality rate ($D_t - D'_t$)

(b) 

(c) 

(d) 

Early Summer

(c) 

(d) 

Summer

(e) 

(f) 

Water flow ($m^3 \cdot s^{-1}$) vs Temperature ($^\circ C$)
Table A2.1: Within-season forecasts of recruitment (R'), target escapement (E'), and actual (i.e. realized) recruitment (R), and realized escapement (E) for four run-timing groups of the Fraser River: Early Stuart, Early Summer, Summer, and Late Summer from 1986 through 2003. Within-season estimates were the latest ones made during the fishing season, when managers could still adjust fishing regulations in order to achieve targets. Realized values are those actually observed after the end of the fishing season. Data were obtained from Fisheries and Oceans Canada and the Pacific Salmon Commission (Table 2.1).

Early Stuart run-timing group

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<th>Return year</th>
<th>Within-season Forecast of recruitment, R'</th>
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**Early Summer run-timing group**

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100
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**Summer run-timing group**

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**Late Summer run-timing group**

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<td>Escapement, E</td>
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<td>escapement, $E'$</td>
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Preface to Chapter 3

In Chapters 1 and 2, I quantified two sources of uncertainties that have only been incompletely considered by previous researchers when forecasting recruitment and evaluating management options. Specifically, in Chapter 2, I characterized the magnitude and patterns in deviations that occur between target and realized mortality rates in fisheries for sockeye salmon on the Fraser River, British Columbia. I found that the direction of those deviations depended on the abundance of recruits and that patterns in deviations varied among management units. I further derived a method to incorporate those patterns into models that analyze management options. However, because of interactions with other components of the fishery system, such as forecasting uncertainty, it is unknown whether management actions that account for this type of uncertainty will improve managers' ability to achieve stated objectives. For example, control systems (e.g., decision algorithms) that account for these sources of uncertainties may be better able to achieve management objectives in the short term than those that ignore them. However, the stabilization of system attributes over time, such as abundance of spawners, may reduce scientists' ability to accurately estimate parameters of forecasting models, resulting in a possible reduction in performance over the long-term (as in the "dual effect of control").

In Chapter 3, I use a computer simulation model of a fisheries system that includes all major biological and management components and their associated uncertainties to prospectively evaluate the performance of two modifications to current
management practices for sockeye salmon in British Columbia and Alaska that account for forecasting and outcome uncertainties described in Chapters 1 and 2. By accounting for those sources of uncertainties, management may be better able to respond to changing age structure, meet specified targets, and therefore achieve their objectives.
Chapter 3
Uncertainties in Population Dynamics and Outcomes of Regulations in Sockeye Salmon Fisheries: Implications for Management

Abstract

Uncertainties in biological and management components of fisheries systems make it difficult to predict the actual outcomes of management actions before implementing them. We evaluated the performance of two modifications to current management practices for sockeye salmon (*Oncorhynchus nerka*) in British Columbia and Alaska in simulations with a computer model that included all major biological and management components and their associated uncertainties. One modification was designed to increase forecasting accuracy by accounting for long-term trends in age-at-maturity; the other was designed to reduce deviations between management targets and actual or "realized" harvest levels (i.e., "outcome uncertainty") by quantitatively taking into account sources of these deviations. We found that, compared with the status quo, performance (i.e., the ability to achieve stated objectives) was increased by reducing outcome uncertainties, but, in contrast to initial expectations, not by reducing forecasting uncertainty. Annual catches were on average higher and less variable over time when management procedures took outcome uncertainty into account. We suggest that any proposals to improve management practices should be routinely evaluated in simulation models that include all major sources of uncertainty.
Introduction

Uncertainties are pervasive in both biological and management components of fisheries systems. They arise because of variability within each component and complex interactions between components (Peterman 2004). The consequences of proposed management actions are usually unclear because of these uncertainties, which are often only incompletely considered by stock assessment scientists when evaluating the effects of potential management actions. Particularly neglected are uncertainties in long-term trends in life history characteristics and accuracy with which management targets are typically achieved.

Control systems theory was developed in the field of engineering science to design rules that achieve specified objectives. Control systems link objectives with information gathered on the state of the system, in efforts to control the system (i.e., hold an attribute in a designated state) (de la Mare 2005). When applied to fisheries, the action taken by management is adapted according to the current state of the fishery and a decision algorithm (i.e., a control system, such as an escapement goal or harvest rule) in a feedback control loop (Walters 1986), typically aiming to stabilize annual catches and population abundances. The observations and assessment of population status combined with the decision algorithm comprise what is called the management procedure (Butterworth et al. 1997; Kirkwood 1997). We would expect that management procedures that account for uncertainties may be better able to control system attributes to achieve management objectives. However, the ability of such management procedures to achieve objectives is not guaranteed because they often use imperfect models of the
real world (i.e., those that ignore key uncertainties) when assessing population abundance or identifying decision algorithms.

Evaluating management procedures by trying them in the real world and observing what happens can be financially costly and creates biological risks as well as risks for industry and management in terms of unacceptably high probability of low harvests or conservation concerns. Instead, a more practical approach to evaluate management actions is to simulate an entire fisheries system in a computer model including biological and management components (i.e., "management strategy evaluation" Sainsbury et al. 2000) (Fig. 3.1). This differs from other simulation models commonly used in stock assessment because it incorporates the known major components of the system and their associated uncertainties, including biological processes, surveys of abundance, scientific assessment of a stock, subsequent management actions, responses of harvesters, and effects on fish population dynamics. Furthermore, these models attempt to reflect reality in that management actions are based on estimates of the "true" population (i.e., from simulated surveys of abundance), but in contrast to the real world, it is possible to compare the simulated "true" abundance in a computer model with the simulated estimate. Furthermore, when management actions are evaluated prospectively in this way, the degree to which objectives are achieved can be identified, and the relative performances of different strategies can be assessed (de la Mare 1996).

Our research objective was to evaluate performance of two modifications to current management procedures for sockeye salmon (Oncorhynchus nerka) fisheries in British Columbia and Alaska in a computer simulation model. The two modifications accounted for uncertainties in (1) a key life-history parameter in the biological system,
The first modification to current approaches pertains to forecasts of recruitment of sockeye salmon (the numbers of adults that return to fresh water every year and become vulnerable to the fishery) prior to the fishing season. Forecasting models, such as the "sibling model" (Peterman 1982; Wood et al. 1997), are used by industry to help prepare for the fishery and by management to set early-season regulations. This model forecasts the abundance of adult recruits in one age group based on abundance of its siblings in the previous year (from the same brood class). Even though the parameters of the sibling model are updated annually as new data becomes available, this model assumes that the distribution of ages-at-maturity remains constant over time. This assumption has been made by stock assessment scientists despite evidence for long-term increases in mean age-at-maturity for sockeye salmon in British Columbia and Alaska over the last half century (Pyper et al. 1999). In a retrospective analysis, a newly developed version of the sibling model that estimated these long-term trends using a Kalman filter reduced mean-squared forecasting errors and mean percent biases compared with the standard sibling model for several stocks of sockeye salmon in Bristol Bay, Alaska (Holt and Peterman 2004). However, any improvements in forecasting accuracy could potentially be swamped by, for example, large natural variation in recruitment or deviations between
management targets and outcomes from fishing regulations (outcome uncertainty). These sources of uncertainty could reduce or negate any improvements in overall performance. In other words, it is not clear whether this newly developed forecasting model will in practice help managers more accurately achieve management objectives.

The second modification to current management procedures pertains to how escapement goals (the target number of spawners) or target harvest rates are identified by stock assessment scientists. One way is to simulate the fisheries system in a simple computer model and use an optimization routine to search for the escapement goal or state-dependent harvest rule (e.g., Fig. 3.2) that best achieves a given management objective. However, the fact that actual harvest levels often deviate substantially from targets (Bocking and Peterman 1988; Eggers 1992; Wood et al. 1998) is rarely accounted for when comparing performances of various escapement goals or harvest rules in simulation models (McAllister et al. 1999; Kell et al. 2005). Such deviations arise, for example, due to imperfect control that management has over outcomes after issuing fishing regulations, i.e., "implementation uncertainty" or "implementation error" (Rosenberg and Brault 1993; Kell et al. 1999), temporal variability in catchability of fish (Cass et al. 2003) and numbers of fishing vessels, as well as errors in estimating fish abundance (Rosenberg and Restrepo 1994). Intuitively we might expect that management performance could be improved by taking these factors into account prior to setting fishing regulations (e.g., by adjusting escapement goals or harvest rules). However, the extent of these improvements is unknown and it is not even clear that performance would improve at all, because there are examples in which greater contrast (i.e., range) in spawner abundance arising from errors in initial parameter estimates can ultimately
produce better parameter estimates when learning is incorporated into management (Walters 1986). This "dual effect of control" originates from the direct value of harvest combined with an indirect value of information that can result in better parameter estimates during stock assessment and possibly higher long-term harvest (Walters 1986; Alexander et al. 2006). Although the management targets (escapement goals and harvest rules) were not updated annually in our simulation model (i.e., they were "open-loop policies"), we did account for learning in the parameters of the forecasting model. Therefore, instead of increasing overall performance, management procedures that account for outcome uncertainty could reduce contrast in spawner data, less-accurate parameter estimates in the forecasting model, reduced forecasting accuracy, and lower mean annual catch over the long term. We thus conducted simulations to determine the net effect of accounting for forecasting and outcome uncertainty.

**Methods**

**Data**

To parameterize the simulation model for evaluating management procedures, we used previously compiled data on age-specific escapement and total recruitment for two sockeye salmon stocks, Togiak River, Bristol Bay, Alaska, (brood years 1952-1998) (Michael Link, pers. comm., LGL, Alaska Research Associates, 1101 East 76th Ave., Anchorage, AK, 99518, USA), and Chilko Lake, Fraser River (part of the summer run-timing group), British Columbia (brood years 1948-1998) (Michael LaPointe, pers. comm., Pacific Salmon Commission, 600 - 1155 Robson St., Vancouver, B.C., V6E 1B5, Canada). As described in more detail later, parameters for observation errors were estimated from unpublished mark-recapture studies in the Fraser River, British Columbia.
(2001-2004) (Timber Whitehouse, pers. comm., Fisheries and Oceans Canada, B.C. Interior and Area and Science Branch, 985 McGill Place, Kamloops, B.C., V2C 6X6), and from visual surveys in Bristol Bay, Alaska (2004-2005) (Tim Baker, pers. comm., Alaska Department of Fish and Game, 333 Raspberry Road, Anchorage, Alaska, 99518-159). Parameters for outcome uncertainties were estimated from empirical records of target and actual ("realized") escapements for the Togiak River, and from target and realized harvest rates from the summer returning group of Fraser River sockeye salmon (1962-2004) (Holt and Peterman 2006). For Chilko sockeye only, we also used previously compiled data on the proportion of spawners that were effective females (1948-1998) (Michael LaPointe pers. comm.) to convert total spawner abundance to effective female spawners.

For Togiak, estimates of recruitment included the sum of spawning escapement (estimated from tower counts) and catch from commercial fisheries, recreational fisheries, subsistence fisheries, personal use, and test fisheries (Plotnick and Eggers 2004). For Chilko, estimates of recruitment included the sum of spawning escapement (estimated from mark-recapture studies) and catch from marine and in-river commercial fisheries, marine and in-river test fisheries, Fraser River First Nations harvests, recreational fisheries, and en-route mortality (Pacific Salmon Commission 2001).

Current management approaches differ for these two stocks. Togiak sockeye salmon are managed for an escapement goal and Chilko sockeye salmon are in part managed using a harvest rule (as in Fig. 3.2). The two modifications to the status quo management procedure evaluated here (accounting for changing age-at-maturity when forecasting abundance and recognizing outcome uncertainty) affect two separate but
related steps in the management of sockeye salmon (forecasting recruitment before the
ing season and identifying escapement goals or harvest rules). The relation between
these two steps differs between the two example salmon populations according to the
management approach used in each case. As described in detail later, for Togiak,
outcome uncertainty (as defined by the difference between target and realized
escapement for this stock) was calculated from forecasting errors, because for this stock,
the deviation between realized and target escapement has historically been greater in
years when forecasts were less accurate (see "Harvesting and escapement" section
below). In contrast, for Chilko, target harvest rates were calculated from forecasts of
recruitment, a target harvest rule, and other factors. However, unlike Togiak, historical
data for Chilko Lake sockeye show that subsequent outcome uncertainties were
independent of forecasts of recruitment.

For both stocks, because management targets and constraints are not explicitly
documented by managers, we assumed that the objective was to maximize yield under
two constraints: allowing no more than a 10% chance of catch dropping below 10% of
maximum sustainable yield, MSY, and no more than a 10% chance of spawner
abundances dropping below 10% of the spawner abundance associated with maximum
recruitment, SMR, in each year. Although the latter reference point is unusual, it is
consistent with what is used for the Sockeye Salmon Spawning Initiative, a recent study
aimed at revising management procedures for Fraser River fisheries (Cass et al. 2003).
These objectives reflect only one possible trade-off between maximizing catch and
minimizing population declines below conservation and catch thresholds, but they
represent practical benchmarks for comparing performances of management procedures.
The next section describes the simulation model, the two modifications to current management procedures that we evaluated, and various sensitivity analyses.

**Simulation model for evaluating management procedures**

The simulation model used to evaluate management procedures had five components: a population dynamics model, the management procedure itself (which contained an observation sub-model, forecasting sub-model, and decision algorithm), the assessment sub-model (which identified the decision algorithm), the effects of harvesting on the population (including outcome uncertainty), and a performance module (Fig. 3.1). We describe each box in Fig. 3.1 in more detail below.

**Population dynamics model – Box 1 of Figure 3.1**

We simulated the dynamics of numerous hypothetical "true" sockeye salmon populations, incorporating stochastic variability in recruitment, age-at-maturity, and proportion of the escapement that were effective female spawners (the latter for Chilko only). For Togiak, true spawner abundance in brood year \( y \), \( S_y \), was used to calculate true recruitment resulting from that spawning, \( R_y \), using a Ricker model with lognormally distributed, autocorrelated random variation (the latter with a one-year time lag):

\[
R_y = a_R S_y e^{-b_P S_y + \phi_y}, \quad \phi_y = \rho \phi_{y-1} + \nu_y, \quad \nu \sim N\left(-\frac{\sigma_\nu^2}{2}, \sigma_\nu^2\right),
\]
where $a_R$, and $b_R$ are parameters, $\varphi$ describes the stochastic deviations in recruitment, $\rho$ is the autocorrelation coefficient, and $\nu$ represents normally distributed random variation (as in Peterman et al. 2003). Note that the normal deviates are centred around $-\sigma_\nu^2/2$ to ensure that the arithmetic mean of the lognormally distributed recruitment multiplier is equal to one, and is equivalent to Hilborn's (1985) bias correction for parameters of log-transformed relations. For this and all subsequent equations, parameters were estimated from historical data using maximum likelihood methods, unless otherwise stated. See Table 3.1 for a list of parameter values.

For Chilko, we used the slightly more complex, Larkin stock-recruit relation (Larkin 1971; Walters and Staley 1987), because this model captures this stock's cyclic pattern in abundance of recruits that repeats every four years. The abundance of the "dominant" line of Chilko Lake sockeye that occurs every fourth year consistently exceeds that of the subsequent "sub-dominant" line and the following two "off-cycle" lines. The Larkin model calculated abundance of recruits from abundance of spawners in the parent generation, as well as the previous year's spawner abundance, $S_{y-1}$:

\[
R_y = a_L S_y e^{(-b_{L0} S_y - b_{L1} S_{y-1}) + \gamma_x}, \quad \gamma \sim N\left(-\frac{\sigma_\gamma^2}{2}, \sigma_\gamma^2\right),
\]

where $a_L$, $b_{L0}$, and $b_{L1}$ are parameters, and $\gamma$ generates normally distributed random variation.
For both Togiak and Chilko sockeye in the simulation model, the proportions of fish of a given freshwater age that matured at each ocean age were calculated from means of historical recruitment data, plus a time trend to reflect the simulated rate of increase in mean ocean age observed for these stocks over the last 50 years. For example, for Togiak, of the fish that spent two winters in freshwater, the proportion that returned after only two winters in the ocean instead of three declined from 0.53 to 0.39 over this time period (and the proportion that returned after three winters increased from 0.47 to 0.61). Summed over all age classes, the increase in mean age-at-maturity for Togiak was 0.186 years, and for Chilko this increase was much smaller (0.049 years). The simulation model also included natural variation in age-at-maturity according to a multivariate logistic distribution (Schnute and Richards 1995) (see Appendix 3.1). Finally, because a portion of the Chilko females do not spawn successfully, we used data on effective female spawners, as is traditional for Fraser River stocks (Pacific Salmon Commission 2001). For this stock, we calculated the proportion of spawners that were effective females, $q$, and incorporated interannual variability in that proportion using a beta distribution, i.e., $q \sim \text{Beta}(\beta_1, \beta_2)$, where $\beta_1$ and $\beta_2$ were also estimated from historical data (see Appendix 3.1).

**Management procedure – Box 2 of Figure 3.1**

Management actions were simulated in three parts: (1) observations of abundances from commercial catch and surveys of escapement, (2) forecasts of recruitment using the sibling model, and (3) escapement goals or target harvest rates identified using a decision algorithm. We evaluated four different management procedures in the simulation model of the fisheries system in our experimental design.
The first management procedure represented the status quo; the second was a modification of the status quo in that the forecasting model accounted for time-varying parameters associated with age-at-maturity using a Kalman filter technique (the "variable maturity" or VM procedure); the third was another modification to the status quo that adjusted management targets for outcome uncertainties (the "outcome uncertainty" or OU procedure); and the fourth incorporated both of these modifications (the "variable maturity and outcome uncertainty" or VM-OU procedure).

Observation sub-model – Box 2a of Figure 3.1

The observed number of spawners, \( S_0 \), in year \( t \), was calculated from the true number of spawners, \( S_t \), with multiplicative, lognormally distributed random variability (as in Cass et al. 2003),

\[
S_{0,t} = S_t e^{\kappa_t}, \quad \kappa \sim N\left( -\frac{\sigma_\kappa^2}{2}, \sigma_\kappa^2 \right).
\]

These random deviations represent both errors in visual observations of spawner abundance (e.g., for tower counts of Togiak sockeye salmon) and sampling variability. We assumed catch was known perfectly because errors due to stock misidentification are negligible for the Togiak and Chilko sockeye stocks, as assessed from analyses of scale patterns and locations of catch that are separate from other stocks (T. Baker pers. comm; T. Whitehouse pers. comm.). In addition, we assumed that ages-at-maturity and proportion of spawners that were effective females were known perfectly.
To properly parameterize natural process variation in recruitment (\(v\) or \(\gamma\), depending on the stock) and observation error (\(\kappa\)), we needed to isolate variability from those two specific sources from the total observed variability in recruitment. Historical recruitment data used to estimate parameters in equations 1 and 2 contained both natural process variation and observation error; hence recruitment calculated from these equations also contained both \(v\) or \(\gamma\), and \(\kappa\). To separate the various contributors to variability, we first set the standard deviation in observation error of spawner abundance, \(\sigma_K\), to 0.02, the mean estimate for observation error in visual surveys of escapement for Togiak in 2004 and 2005 (T. Baker pers. comm.). This corresponded to a relatively low level of observation error for Chilko, as shown by mark-recapture studies on ten Fraser River stocks, where annual estimates of \(\sigma_K\) ranged from 0.02 to 0.19 between 2001 and 2004 (T. Whitehouse pers. comm.). In a simple simulation model, we then estimated the standard deviations of the natural process variation, \(\sigma_v\) and \(\sigma_\gamma\) (for equations 1 and 2 for Togiak and Chilko, respectively), which, in combination with these observation errors in a simulation, resulted in standard deviations in total recruitment equal to those observed historically (\(\sigma_t\)). Note that those variances were not additive (i.e., \(\sigma^2_t \neq \sigma^2_v + \sigma^2_\gamma\)), indicating that the sources of variability were not independent. We found that the total combined uncertainty (\(\psi\)) was swamped by large natural variability in recruitment. Specifically, the standard deviation in natural process variation was equal to 99% of the standard deviation in total combined recruitment for both stocks (i.e., \(\sigma_v = 0.99\sigma_t\) and \(\sigma_\gamma = 0.99\sigma_t\)). In a sensitivity analysis, we increased the magnitude of observation errors to \(\sigma_K = 0.19\), which corresponded to standard deviations in natural process variation that were
equal to 94% of the standard deviation in total observed recruitment for Togiak ($\sigma_v = 0.94\sigma_y$), and 90% for Chilko ($\sigma_v = 0.90\sigma_y$).

*Forecasting sub-model – Box 2b of Figure 3.1*

As described in Table 3.2, we evaluated management procedures that used two different forecasting models. For the status quo and OU procedure, we used a standard sibling model that assumed constant parameters over time to forecast recruitment. For the VM and VM-OU procedures, we modified the standard linear sibling model to account for changes in age-at-maturity using a Kalman filter (see Holt and Peterman 2004 for a description of these two parameterizations of the sibling model). To forecast the youngest age-class for each freshwater age, we simply used recruitment in the previous generation, instead of the sibling model (as in Haeseker et al. 2006).

We expected that when simulated over many years and Monte Carlo trials, the management procedure that forecasted recruitment using a sibling model that accounted for long-term trends in age-at-maturity (the VM procedure) would have lower forecasting errors, improved accuracy of achieving management targets, and therefore higher mean annual catch than the status quo management procedure that used a sibling model with constant parameters (which produces a constant distribution of ages-at-maturity).

*Decision algorithm – Box 2c of Figure 3.1*

Annual management targets were then chosen based on decision algorithms that were identified in the Assessment sub-model (Box 3). For Chilko, management targets were identified from target harvest rules, and for Togiak, escapement goals were used.
Assessment sub-model – Box 3 of Figure 3.1

In the assessment sub-model, we identified two management decision algorithms for each stock, one that ignored outcome uncertainty and one that accounted for it. As described in Table 3.2, for the status quo and VM procedures, we used management decisions algorithms that ignored outcome uncertainty, and for the OU and VM-OU procedures, we used algorithms that accounted for it. This sub-model was simulated separately from the model used to evaluate management options (i.e., it was run once for each management procedure in our experimental design). Specifically, in the assessment sub-model for Togiak, we simulated the biological system and management actions using various escapement goals to identify those that would best meet management objectives among those considered, i.e., maximize yield while maintaining spawner abundance and catch above minimum thresholds stated previously. This process was repeated over 1000 Monte Carlo trials. For Chilko, instead of identifying escapement goals, we identified state-dependent harvest rules. Specifically, the target harvest rates, $h_t$, in year $t$, were calculated as a function of forecasts of recruitment, $R'_t$, using the following target harvest rule:

\[
(4) \quad h'_t = c(1 - \exp(d(m - R'_t))) ,
\]

where $c$ is the maximum harvest rate (proportion) at high recruitment, $d$ is a shape parameter, and $m$ is the protection level of spawner abundances below which harvest rates are zero.
This assessment sub-model was a modified version of the overall simulation model with four simplifications. These four simplifying assumptions, described below, reflect managers' current considerations when choosing decision algorithms. However, in contrast to our assessment sub-model, rigorous quantitative analyses are currently not performed when identifying decision algorithms for these stocks.

1. Assumptions about recruitment

For Togiak, the assessment sub-model assumed all recruits were age 5 years at maturity, because from 1956 through 2001, on average 67% of recruits were age 5. For Chilko, the assessment sub-model assumed that adults returned at age 4 and age 5 because ecological interactions between fish that return in subsequent years, in part due to differential age-at-maturity, are an important component of cyclic population dynamics of Chilko Lake sockeye salmon (Ricker 1997). The proportions of recruits that were age 4 in year \( t \), \( p_{4,t} \), were calculated from the proportion of spawners that were age 4 in the parent generation, \( p'_{4,t} \) (Walters and Woodey 1992), using the following logistic model:

\[
(5) \quad p_{4,t} = \left(1 + f_0 f_1 p'_{4,t}\right)^{-1} + \tau, \quad \tau \sim N\left(0, \sigma^2\right),
\]

where \( f_0 \) and \( f_1 \) are parameters, and \( \tau \) represents normally distributed random variation. The dependent variable, \( p_4 \), was always between zero and one.

2. Assumptions about trends in age-at-maturity

In contrast to the overall simulation model, we did not incorporate long-term trends in age-at-maturity in the assessment sub-model.
3. Assumptions about forecasts of recruitment

In the assessment sub-model, we did not explicitly simulate forecasts of recruitment. Instead, for Togiak we simulated errors in forecasts only, and for Chilko, we did not simulate forecasts or errors in forecast because outcomes of fishing regulations were independent of those factors. Specifically, for Togiak, we simulated a stock assessment procedure that took into account forecasting error from an empirically based relation with realized recruitment. Our analyses of historical data for that stock shows that forecasts of recruitment underestimate realized recruitment when recruitment is large and overestimate realized recruitment when recruitment is small (Fig. 3.3). Therefore, we calculated forecasting error (the natural log of the ratio of forecasts of recruitment, $R'$, to realized recruitment, $R_t$) from realized recruitment:

\[
\log_e \left( \frac{R'_t}{R_t} \right) = \eta_0 + \eta_1 \log_e (R_t) + \varphi_t, \quad \varphi \sim N(0, \sigma^2),
\]

where $\eta_0$ and $\eta_1$ are parameters, and $\varphi$ represents normally distributed variability in forecasting errors. We used log-transformed abundances because of empirical and theoretical support for lognormally distributed variation in recruitment (Peterman 1981; Hilborn and Walters 1992).

4. Assumptions about outcome uncertainties

In the assessment sub-model, we first assumed that escapement goals and harvest rules were implemented precisely and without bias (for the status quo and VM procedure, Table 3.2). In a subsequent analysis, we simulated imprecision and bias as described for
the overall simulation model (for the OU and VM-OU procedures, Table 3.2). As a result of running the assessment sub-model under these two circumstances, we identified two escapement goals for Togiak and two harvest rules for Chilko, one that ignored outcome uncertainty (an escapement goal, \(E'\), and a harvest rule with parameters, \(c, d, \) and \(m\) (equation 6), for Togiak and Chilko, respectively) and one that accounted for outcome uncertainty (an escapement goal, \(E'^*\), and a harvest rule with parameters \(c^*, d^*, \) and \(m^*\), for Togiak and Chilko, respectively) (parameter values in Table 3.1).

When simulating the fishery over years and Monte Carlo trials, we expected that the procedure that accounted for outcome uncertainty when identifying target escapements or harvest rules (the OU procedure) would have higher mean abundance and annual catch and hence better achieve management objectives than the procedure that ignored this source of uncertainty. Furthermore, we expected the management procedure that used both modifications (the VM-OU procedure) would also better achieve management objectives than the status quo.

**Harvesting and escapement – Box 4 of Figure 3.1**

In the fourth component of the simulation model, we simulated uncertainty in the outcomes of applying management targets. For Togiak, we simulated bias in realized escapement from the target using an empirical relation with forecasting errors. Specifically, historical data showed that forecasting errors (defined as \(\log_e(R'_i/R_i)\), where \(R'_i\) is the forecast of recruitment and \(R_i\) is realized recruitment) were negatively related to deviations between realized and target escapements (\(\log_e(E_i/E'_i)\), where \(E_i\) is realized escapement, and \(E'_i\) is escapement goal) (Fig. 3.4, \(p<0.001\)). For example, when pre-season forecasts under-estimate the actual, realized recruitment (i.e., to the left of 1.0 on
the X-axis of Fig. 3.4), more fish return to spawning grounds than targeted by the escapement goal. The opposite occurred (less fish escaped the fishery than targeted) in only 2 out of 21 years (i.e., 2 points fell below 1.0 on the Y-axis of Fig. 3.4). Therefore, we calculated escapement deviations (i.e., \( \log_e(E/E') \)) from forecasting errors, \( F_i \):

\[
(7) \quad \log_e \left( \frac{E_i}{E'_i} \right) = \lambda_0 + \lambda_1 F_i + \delta_i, \quad \delta \sim N(0, \sigma^2),
\]

where \( \lambda_0 \) and \( \lambda_1 \) are parameters, and \( \delta \) represents normally distributed variability in \( \log_e(E/E') \).

For Chilko, there is little empirical evidence for bias in realized harvest rates from targets for the summer returning group of the Fraser River, so we simulated imprecision in the outcome from target harvest rules without bias (Holt and Peternan 2006). We included beta-distributed random variability in realized harvest rates to simulate imprecision in the outcome, constraining the harvest rates, \( h_i \), to between zero and one:

\[
(8) \quad h_i = \text{Beta}(\beta_1, \beta_2),
\]

where \( \beta_1 \) and \( \beta_2 \) are parameters of the beta distribution, which are a function of the target harvest rate from equation 4 and the variance in realized harvest rates around the harvest rule (\( \sigma^2 = 0.031 \), estimated from historical observations) (see Appendix 3.1).
Historical escapement data used to estimate the parameters of equations 7 and 8 contain observation error as a component of outcome uncertainty. For the simulation model that evaluated management procedures, we separated this source of variability (since it was already simulated in the Observation sub-model) using the same method as described in the "Population dynamics model" sub-section. However, for the assessment sub-model, this separation was not necessary since observations of abundance were not simulated independently of natural variability in recruitment (i.e., we did not include a separate observation sub-model). For Togiak, the standard deviation in remaining outcome uncertainty ($\sigma_\delta$) was 94% of the standard deviation in total observed outcome uncertainty, and this value was 99% for Chilko. When we increased observation errors in a sensitivity analysis ($\sigma_x = 0.19$, instead of $\sigma_x = 0.02$), these values ($\sigma_\delta$) were 70% and 88% for Togiak and Chilko, respectively.

**Performance module – Box 5 in Figure 3.1**

To evaluate how well the management procedures achieved objectives, the following metrics were averaged over each 50-year simulation period:

1. mean annual catch,
2. proportion of years when catch dropped below 0.1·$MSY$,
3. proportion of years when escapement dropped below 0.1·$SMR$ (10% of spawner abundance at maximum recruitment),
4. median percent change in catch between successive years,
5. root-mean-square forecasting error,
(6) mean absolute forecasting error, and

(7) mean of the absolute deviations between target and realized escapements or harvest rates.

In addition to those attributes described by the objective (performance metrics 1-3), harvesters may prefer reductions in variability of catch among years (captured by metric 4, referred to as catch variability), whereas scientists and managers may be more interested in reductions in forecasting errors (captured by metrics 5 and 6, referred to as RMS forecasting error and bias of forecast, respectively) and deviations between management targets and outcomes (captured by metric 7, referred to as outcome deviation). Simulations were repeated over 500 Monte Carlo trials (each over 50 years), the number necessary to stabilize output metrics (less than in the assessment sub-model because this simulation to evaluate management options included iterations over years to allow for long-term trends in age-at-maturity). Performance metrics were averaged over all trials. Because the same series of random numbers were used when simulating the performance of each management procedure, any differences in performance were due to inherent properties of the procedures.

Sensitivity analyses

We evaluated the sensitivity of the rank order of the four management procedures (Table 3.2) to five scenarios: (1) larger increases in age-at-maturity than those previously observed in the field data, (2) sinusoidal time trends in age-at-maturity, (3) constant age-at-maturity, (4) larger magnitudes of observation errors than typically observed for these stocks relative to natural variation in recruitment and outcome uncertainty, and (5)
alternative forms of the stock-recruitment relation in the population dynamics model. For the latter scenario, we specifically examined a model that included depensatory mortality at low population sizes (Appendix 3.2), and for Chilko, a scenario that did not incorporate cyclic patterns in recruitment because for that stock it is unknown whether the observed cyclic dynamics are intrinsic or just a result of cyclic harvesting pressure (Walters and Staley 1987; Cass and Grout 2006). For this latter scenario, we used equation 1, with appropriate parameter values, instead of equation 2 to calculate recruitment.

Finally, to examine the potential consequences of adjusting harvest regulations for outcome uncertainties when the latter do not exist or are over-estimated compared with the actual level of uncertainty in the "true" population, we omitted outcome uncertainty from the overall simulation model and calculated the difference in mean annual catch between cases that used either the OU procedure or the status quo method. In other words, we evaluated the potential reductions in performance when outcome uncertainty was accounted for but did not occur in the "true" population, compared with the status quo procedure.

Results

Togiak

We first present results for Togiak, then those for Chilko. As expected, compared with the status quo management procedure, mean annual catch for Togiak sockeye salmon was higher for the OU and VM-OU procedure, i.e., the ones that accounted for outcome uncertainties (Fig. 3.5a). Note, the escapement goal that accounted for outcome
uncertainty (132 660 spawners) was lower than the goal that ignored it (179 176 spawners) to compensate for the observed trend in over-escapement (Fig. 3.4), arising from incentives for managers to meet or exceed escapement goals (Hilborn 2006). Compared with the status quo management procedure, mean annual catch tended to be higher for the VM procedure, i.e., that which accounted for long-term trends in age-at-maturity, but this difference was small (0.3% increase in catch). For this and the subsequent five plots in Fig. 3.5, management procedures are aligned on the Y-axis according to their rank in mean annual catch (Figs. 3.5a and d). To show trade-offs between mean annual catch and other performance metrics, values for additional metrics are included on the plots 3.5a-f scaled according the bottom X-axis.

The mean proportion of years when annual catch dropped below 0.1·MSY was consistently low for all four procedures (<0.07, i.e., less than the 0.1 stated in the management objective). This was also true for the mean proportions of years in which spawning abundance dropped below 0.1·SMR (<0.001, i.e., less than the 0.1 management objective). Because sockeye spawning stocks of Bristol Bay, Alaska are often compared with their spawning abundance at unfished equilibrium,  $S_{eq}$, we further calculated the mean proportion of years when spawning abundance dropped below 0.1·$S_{eq}$. Again, this value was low for all procedures (<0.001).

For Togiak, the rank orders of management procedures were consistent across the remaining performance metrics. The management procedures that had the highest mean annual catch also had the lowest catch variability (i.e., performance improved on both metrics) (Fig. 3.5a). In contrast to our initial expectations, the RMS forecasting error was not consistently smaller for the VM procedure compared with the status quo, even though
long-term trends in age-at-maturity occurred in the simulated "true" population (Fig. 3.5b). However, compared with the status quo, the RMS forecasting error was consistently smaller for the OU procedure and the combined VM-OU procedure (Fig. 3.5b). Forecasts for the OU procedure and VM procedure were also less negatively biased than the status quo and the absolute magnitude of bias was reduced (not shown). As expected, compared with the status quo, the outcome deviations were lower (i.e., performance improved) for the OU and VM-OU procedures (by approximately 29,900 spawners, or 6% of mean annual recruitment) (Fig. 3.5c). That metric was also lower for the VM procedure compared with the status quo, but only by 855 spawners or approximately 0.2% of mean annual recruitment.

Overall for Togiak, the OU procedure outperformed the VM procedure and the status quo on all performance metrics (Fig. 3.5). The VM-OU procedure performed similarly, outperforming the VM procedure alone and the status quo.

**Chilko**

These results were similar to Togiak; mean annual catches were higher for the OU procedure compared with the status quo, but this increase was small (1,944 fish or approximately 0.1% of mean annual recruitment) (Fig. 3.5d). In contrast to our initial expectations, compared with the status quo, mean annual catches were lower for the VM procedure and the VM-OU procedure (by 25,967 and 67,269 fish, or approximately 2% and 5% of mean annual recruitment, respectively). As was the case for Togiak, the mean proportion of years when annual catch of Chilko Lake sockeye dropped below 0.1·MSY and the mean proportion of years when spawning abundance dropped below 0.1·SMR
were consistently low for all procedures (<0.07 and <0.06, respectively, i.e., less than the 0.1 in the management objective).

The time series of catches were least variable for the OU procedure, followed by the status quo, VM-OU, and VM procedures in order of increasing variability (Fig. 3.5d). However, the range of % change in catch across those cases was small (<4%). Counter to expectations, the RMS forecasting error of the VM, OU, and VM-OU procedures were larger (i.e., had worse performance) when compared with the status quo (Fig. 3.5e). In contrast, the three modified management procedures had forecasts with smaller positive bias by 4 to 41% compared with the status quo (not shown). Outcome deviations were 1% lower for the OU procedure compared with the status quo, but the opposite trend was true for the VM procedure (Fig. 3.5f).

Although the OU procedure outperformed the VM procedure and the status quo on most performance metrics for Chilko, these improvements were small compared with those observed for Togiak (e.g., 1% increase in mean annual catch for the OU procedure over the status quo for Chilko, compared with 11% increase for Togiak). The status quo outperformed the remaining two management procedures on all other performance metrics for Chilko Lake sockeye.

Sensitivity analyses

For both stocks, the rank order of management procedures was not sensitive to most changes made in our sensitivity analyses. First, there was no change in ranking when evaluating different time trends in age-at-maturity (annual increases twice as large as those observed historically, sinusoidal changes, and no long-term trends). For
example, for Togiak sockeye salmon, the relatively small increases in catch for the VM procedure over the status quo management procedure occurred regardless of the trends in age-at-maturity in the "true" population (<1% increase over the status quo for both cases when trends occurred and when they were absent). Furthermore, for both stocks, the rank order of management procedures was not sensitive to increases in the magnitude of observation error relative to natural variation in recruitment and outcome uncertainty, or to an alternative structural form of the stock-recruit relation that incorporated depensatory mortality at small abundances.

However, as opposed to our initial results for Chilko Lake sockeye salmon where modifications to the status quo management procedure resulted in reduced mean annual catch in some cases, when we assumed that recruitment patterns were non-cyclic rather than cyclic in the "true" population, the mean annual catch for the VM, OU, and VM-OU procedures exceeded that of the status quo (by 8%, 2%, and 5%, respectively). Also, compared with the status quo, forecasts of recruitment from these management procedures were less biased (the mean absolute forecasting errors were reduced by between 22 and 78%) and outcome deviations were lower (by between 0.2 and 2.0%) (i.e., performance improved on both metrics).

In a final sensitivity analysis, we considered the scenario where management targets were achieved exactly. Although this scenario is unlikely, it represents a lower bound on the magnitude of outcome uncertainty. For management procedures that "mistakenly" accounted for outcome uncertainties, i.e., when we assumed that escapement goals or target harvest rates were going to be achieved exactly in the simulated "true" population but management acted as if outcomes were uncertain, the
mean annual catches were lower than the status quo management procedure by 2% and 5% for Togiak and Chilko, respectively. In other words, when managers incorrectly assumed that management decisions were implemented inaccurately, the annual yields were 2 to 5% lower than when the status quo management procedure was applied.

Discussion

Although accounting for key biological and outcome uncertainties in stock assessments may a priori seem justifiable for improving the assessment and management of sockeye salmon, improvements in any one component of our simulated fisheries system in our simulation did not necessarily enhance performance overall. For example, accounting for outcome uncertainty (the outcome uncertainty or OU management procedure) improved management performance, but accounting for forecasting uncertainty (the variable maturity, or VM procedure) did not. Improvements from the OU procedure represented annual harvests that were maintained at a relatively high level. Although stabilization of catch may have been associated with stabilization of recruitment and reduced learning about the parameters of the sibling model, any reductions in performance (mean catch) because of poor parameter estimates were overshadowed by improvements by accounting for outcome uncertainty.

Performance of the OU procedure

A control system (i.e., decision algorithm) aims to maintain attributes of a system (e.g., catch and spawning escapement) at levels required to achieve objectives. Compared with the status quo, the management procedure that accounted for outcome uncertainty was better able to do this for both Togiak and Chilko, increasing catch and
also reducing interannual variability in catch and total recruitment (the latter not shown). For Togiak, forecasting accuracy also improved for the OU procedure because for this stock, forecasting accuracy tended to be highest for average levels of recruitment, and lowest at extremely large and small recruitment (Fig. 3.3).

The increase in catch for management procedures that correctly accounted for outcome uncertainty (when present) can be explained, at least for Togiak, by the bias that causes realized escapements to exceed targets, as evident in the historical record for that stock (Fig. 3.4). When target escapements are adjusted downwards to compensate for that bias, a larger proportion of the recruitment can be harvested (so mean annual catches increase). Not surprisingly, outcome deviations were lower for this management procedure compared with the status quo. Therefore, by ignoring differences between management targets and outcomes when they exist, managers of Togiak sockeye salmon may be inadvertently foregoing some long-term catch.

Similarly, for Chilko, the management procedure that accounted for outcome uncertainties was better able to maintain high levels of catch. However, in contrast to Togiak, for Chilko sockeye, the outcome of harvest rates was unbiased both in the historical record and in the "true" population in our simulation model of the fisheries system, so improvements in mean annual catch were due only to accounting for the imprecision in achieving target harvest rates, and those benefits were relatively small.

When we incorrectly assumed in the assessment sub-model (which generated the harvest rule) that recruitment of Chilko Lake sockeye exhibited cyclic dominance as opposed to being non-cyclic, the improved performance of the OU procedure compared with the status quo can be explained by the assessment sub-models' incorrect description
of that stock's productivity. For both the OU procedure and status quo, the harvest rule that mistakenly assumed cyclic dominance also assumed the stock was more productive than the "true" population because cyclic dynamics were associated with higher average recruitment than non-cyclic dynamics. For the status quo, the incorrect assumption about the increased productivity (due to incorrect assumption about cyclic dominance) resulted in overexploitation of the stock and relatively low mean annual catches. In contrast, for the OU procedure, harvest rates were adjusted downwards compared with the status quo, therefore the increased exploitation when we incorrectly assumed cyclic dominance actually resulted in relatively high mean annual catch.

In the above analysis, we assumed that the outcomes of regulations differed from targets in the "true" population; however, the magnitudes and even the occurrence of these differences are not always evident. When we instead assumed that outcomes exactly matched targets but managers (incorrectly) accounted for outcome uncertainty, for Togiak, the reduction in mean annual catch over the status quo was much less than the increase in catch when the opposite occurred (i.e., when managers correctly assumed that outcomes differed from targets). That is, when management procedures were implemented exactly in the "true" population (i.e., when outcome uncertainty did not exist), the potential losses associated with mistakenly adjusting escapement goals (i.e., overestimating outcome uncertainty) were small (2% reduction in mean annual catch) compared with the gains from correctly adjusting escapement goals when they were, in fact, implemented with uncertainty in the "true" population (11% increase in mean annual catch). However, this was not the case for Chilko; the losses from mistakenly adjusting target harvest rates (5% reduction in mean annual catch) were larger than the gains
associated with adjusting target harvest rates when they were, in fact, implemented with uncertainty in the "true" population (0.1% increase in mean annual catch). In other words, the changes in performance were asymmetric. If managers are unsure whether outcome uncertainty exists (assigning equal probability to its occurrence and absence), for Togiak the highest expected catch is achieved from adjusting management targets nonetheless, but not for Chilko.

Performance of the VM procedure

For Togiak, the VM procedure that used the Kalman filter technique to forecast recruitment improved forecasting accuracy only slightly over the status quo, and for Chilko, these forecasts were consistently less accurate. The latter inaccurate forecasts were, in part, due to the tendency of the forecasting model that used a Kalman filter to track random variability in age-specific recruitment ("noise") and the cyclic patterns in Chilko recruitment, instead of long-term trends (i.e., "process" variability), which were relatively small in magnitude. This resulted in large interannual variability in parameters of the Kalman filter forecasting model, and hence relatively inaccurate forecasts of recruitment and reductions in mean annual catch. Furthermore, the rank order of management procedures was independent of long-term trends in age-at-maturity in the "true" population, further suggesting that the VM procedure was unable to appropriately capture these trends for Chilko Lake sockeye salmon.

We initially suggested that because of the dual effect of control, increases in contrast in observed data could result in improved performance if learning about the parameters of the sibling model resulted in higher catches over the long term. For the standard sibling model in the case of Chilko Lake sockeye, forecasting accuracy
improved when outcome uncertainties were ignored (i.e., the status quo was better than the OU procedure). This result may have been due to increased variability in recruitment and catch when outcome uncertainty was ignored, resulting in larger contrast in simulated observed data and therefore better estimates of parameters of the forecasting model. However, any increases in forecasting accuracy did not result in increased mean annual catch (i.e., the indirect effects of learning were not important to overall performance). For Togiak, management procedures that ignored outcome uncertainties (and reduced management control) did not allow for greater learning about the parameters of the forecasting model. In fact, forecasting accuracy decreased when escapement was very high or low for that stock.

Nevertheless, larger improvements in forecast accuracy with other future modifications to the sibling or other models that were not considered here may result in improved mean annual catch. For example, for Chilko, the potential increase in mean annual catch over the status quo with a completely accurate forecasting model was 13%. However, given large natural variability in population dynamics, ecological interactions within and among stocks, and climate effects, such improvements in forecasting accuracy are unlikely to occur in the foreseeable future.

Comparison with previous research: outcome uncertainty

Despite repeated recommendations to incorporate implementation uncertainties (one component of outcome uncertainties) into simulation models when evaluating management options for various fisheries (e.g., Rice and Richards 1996; McAllister et al. 1999), fisheries models that include them are the exception and not the rule. Those previous studies that did incorporate outcome uncertainties into simulation models
assumed that those differences are either deterministic (i.e., a fixed bias) (Kell et al. 1999) or if stochastic, that they are unbiased with respect to the target (e.g., Johnston et al. 2000). In the few cases where both imprecision and bias have been included, they have been parameterized qualitatively either using empirical data (Kell et al. 1999; Peterman et al. 2000) or expert opinion (Kell et al. 1999; Nowlis and Bollermann 2002). Furthermore, those previous studies have not evaluated management procedures that specifically account for these deviations (Carlson et al. 1999; Peterman et al. 2000), with one exception (Cass et al. 2003). In that study, Cass et. al. (2003) identified harvest rules for two Fraser River stocks that accounted for uncertainty in achieving management targets assuming that differences between realized harvest rates and targets were characterized by a random normal deviation around the target parameterized by expert judgment. Similar to our results for Chilko, those authors found that accounting for imprecision in this type of uncertainty resulted in relatively small changes in the form of the target harvest rule (Cass et al. 2003). However, in contrast to our study, Cass et al. (2003) did not take bias into account when evaluating management options.

**Comparisons with previous research: evaluating forecasting models**

Several studies have quantitatively evaluated the performance of a wide range of forecasting models for sockeye salmon (Bocking and Peterman 1988; Noakes et al. 1990; Wood et al. 1997; Adkison and Peterman 2000; Peterman et al. 2003; Haeseker et al. 2006), but none has evaluated sibling models that use a Kalman filter, let alone by using a simulation model of an entire fishery system. A common assumption in these studies is that improvements in forecasting accuracy will result in higher yields and economic gains, but our results show that this is not necessarily the case.
One example of a study that supported the assumption that more accurate forecasts improve management performance used a computer simulation model to evaluate another type of forecasting model, the stock-recruit relation (Peterman et al. 2000). This study compared parameter estimation methods that accounted for long-term trends in productivity using a Kalman filter to methods that did not. In contrast to our results, they found that the forecasting model that used a Kalman filter had higher cumulative catches compared with the standard forecasting model. However, Peterman et al. (2000) used additive, normally distributed deviates between target and realized escapement in the "true" population, whereas we used multiplicative, lognormally distributed deviates characterized by a relatively large standard deviation (\(\sigma = 0.31\) for the normal distribution transformed from natural logarithms, compared with \(\sigma = 0.20\) for Peterman et al. (2000)). In addition, Peterman et al. (2000) had a different formula for bias in what they called implementation error (i.e., imperfect control that management has over implementing harvest policies) than the formula we used for our metric, outcome uncertainty. This difference in model form resulted in higher overall performance of their forecasting model compared with ours. Furthermore, they simulated much larger temporal changes in the variable being estimated (the Ricker \(a\) parameter) than we did for our variable (the Y intercept of the sibling model).

Limitations

Interpretations of our results are affected by at least three limitations. First, large stock-to-stock differences in population dynamics and management approaches preclude generalizing our results to other sockeye salmon stocks with confidence, let alone to other species. However, it is likely that the OU procedure will only outperform the status quo
management procedure when such uncertainty in the "true" population creates a bias between targets and realized outcomes rather than just imprecision alone. In addition, it is unlikely that the VM procedure will perform well in other sockeye salmon stocks that show large among-year variability in recruitment (e.g., for cyclic stocks such as Chilko).

Second, within-season forecasts of recruitment often replace pre-season forecasts fairly early in the fishing season for sockeye salmon as more accurate estimates of abundance become available, but these within-season forecasts were not explicitly incorporated into our management procedures. However, within-season forecasts were implicit in our calculation of outcome uncertainty in the "true" population (equations 7 and 8) because the parameters of these relations were estimated from empirical data for which management agencies used within-season updates of recruitment. The complexity of within-season management precluded us from explicitly simulating these steps in our management procedure.

Third, although we did update the forecasting model with annual recruitment data, we did not incorporate learning into the assessment sub-model that identified desired harvest rules and escapement goals. Further improvements in performance may be expected with periodic updates of the assessment sub-model, especially if the management procedure allows for sufficient contrast in recruitment data to more accurately estimate parameters. In the current analysis, our objective was simply to evaluate any improvements from accounting for outcome uncertainty in an "open-loop" policy, though the analysis could be extended to evaluate such "closed-loop" policies.
Management recommendations

This study shows that assessment methods with more realistic assumptions about the presence and nature of uncertainties may or may not result in improved performance when put in the context of an entire fishery. Several previous authors (e.g., de la Mare 1998; Cooke 1999) have suggested that proposed management procedures (including forecasting and stock assessment models) be evaluated in as comprehensive of a simulation model as possible prior to applying them to the fishery (as demonstrated for other fisheries such as North Sea plaice (Kell et al. 1999) and the Australian gemfish fishery (Punt and Smith 1999), to name two examples of many). By simulating the fishery over various future scenarios, such evaluations avoid the small sample sizes involved in assessing performance using retrospective analyses or real-time experimentation. Computer simulation models also allow for learning about, for example, the causes of population collapses because the state of the "true" stock is known (Cooke 1999). Furthermore, as suggested by many of the authors listed above, sensitivity analyses should be used to evaluate management procedures in the face of uncertainties in underlying models describing system dynamics (McAllister et al. 1999; Punt and Smith 1999). For example, Cooke (1999) recommends considering different assumptions about spatial structure of stocks, trends in productivity over the medium and long-term, interactions among species, as well as management scenarios with misleading data (i.e., observations and/or assessments that do not reflect the state of the "true" population”). In this way, managers can choose procedures that are relatively robust to structural uncertainty. Finally, stock-specific differences in population dynamics and management approaches necessitate evaluations of these procedures that are also stock-specific.
Acknowledgments

We thank the Alaska Department of Fish and Game and Fisheries and Oceans Canada for kindly providing the data for sockeye salmon. Statistical advice was provided by Bill de la Mare and Sean Cox. Funding for this study was provided by grants from the Natural Sciences and Engineering Council of Canada, the Canada Foundation for Innovation, the British Columbia Knowledge Development Fund, and Fisheries and Oceans Canada, awarded to R. M. Peterman.

References


### Tables

Table 3.1 Parameter values used in the simulation model to evaluate management procedures and/or in the assessment sub-model to identify management targets.

<table>
<thead>
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<th>Parameter value</th>
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<th>Chilko</th>
<th>Equation number</th>
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Table 3.2 Description of the four management procedures that we evaluated in the simulation model.

The VM procedure is a modification to the status quo that accounts for forecasting uncertainty. The OU procedure is a modification to the status quo that accounts for outcome uncertainty. The VM-OU procedure accounts for both variable maturity and outcome uncertainty. The second and third columns describe modifications to two components of the management procedure: the forecasting sub-model and assessment sub-model.

<table>
<thead>
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<th>Components of management procedure</th>
<th>Forecasting sub-model</th>
<th>Assessment sub-model</th>
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<td>Assumes that management targets are implemented exactly</td>
</tr>
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<td>Time-varying parameters accounted for using a Kalman filter technique</td>
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</tr>
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<td>OU procedure</td>
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<td>Management targets adjusted for outcome uncertainty</td>
</tr>
<tr>
<td>VM-OU procedure</td>
<td>Time-varying parameters accounted for using a Kalman filter technique</td>
<td>Management targets adjusted for outcome uncertainty</td>
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</table>
Figures

Figure 3.1    Simplified outline of the simulation model used to evaluate management procedures for sockeye salmon fisheries in British Columbia and Alaska. Numerous uncertainties are taken into account in each box except the top and bottom ones.
Figure 3.2  Example of a harvest rule used to identify target harvest rates for sockeye salmon fisheries in the Fraser River, British Columbia. PL is the protection level in abundance of recruits below which target harvest rate is zero, and $R_M$ is the maximum observed recruitment. In many salmon fisheries (depending on the management objective), the PL is the escapement goal, or target number of spawners.
Figure 3.3  Relation between realized abundance of recruits, $R_t$, in year $t$, (millions of fish) and forecasting errors ($R_t'/R_t$) plotted on a logarithmic scale, where $R_t'$ is the forecast of recruitment, for the Togiak River sockeye salmon, Bristol Bay, Alaska (1984-2004) ($r^2 = 0.72$, $P < 0.001$). The horizontal dashed line represents a perfect forecast ($R_t'/R_t$ equal to 1.0).
Figure 3.4  Relation between annual forecasting error \( (R_t'/R_t) \) and outcome uncertainty \( (E_t/E_t') \) plotted on a logarithmic scale, where \( R_t' \) is the forecast of recruitment in year \( t \), \( R_t \) is realized recruitment, \( E_t \) is realized escapement, and \( E_t' \) is the escapement goal, for the Togiak River sockeye salmon, Bristol Bay, Alaska (1984-2004) \( (r^2 = 0.65, \ P < 0.001) \). The vertical dashed line represents a prefect forecast \( (R_t'/R_t \ equal \ to \ 1.0) \) and the horizontal dashed line represents an escapement goal that is implemented exactly \( (E_t/E_t' \ equal \ to \ 1.0) \).
Figure 3.5 Trade-offs in performance metrics for sockeye salmon fisheries on Togiak River, Bristol Bay, Alaska (a), (b), and (c) and Chilko Lake, Fraser River, British Columbia (d), (e), and (f) sockeye salmon for four management procedures: outcome uncertainty or OU procedure, forecasting and outcome uncertainty or VM-OU procedure, forecasting uncertainty or VM procedure, and status quo, arranged on the vertical axis according to their rank in mean annual catch. Solid lines and open circles represent mean annual catch; dashed lines and hollow triangles represent median percent change in catch between successive years (i.e., catch variability) on plots (a) and (d), root-mean-square forecasting error for recruitment (i.e., RMS forecasting error) on plots (b) and (e), and mean absolute deviation between target and realized escapement (i.e., outcome deviation) on plots (c) and (f). Error bars represent ± one standard error of the mean result across 500 Monte Carlo trials on all plots except (c), for which the error bars are too narrow to distinguish from the mean. (See following page.)
Appendix 3.1 – Random distributions

Multivariate logistic distribution

We simulated proportions at age \(a\), \(p_a\), in year \(t\), using a multivariate logistic distribution:

\[
P_{a,t} = \frac{e^{x_{a,t}}}{\sum_{A=1}^{A} e^{x_{a,t}}} ,
\]

where,

\[
x_{a,t} = \log_e(\bar{p}_{a,t}) + \omega \varepsilon_{a,t} - \frac{1}{A} \sum_{A=1}^{A} \left[ \log_e(\bar{p}_{a,t}) + \omega \varepsilon_{a,t} \right],
\]

\(A\) is the oldest age class, \(\bar{p}_{a,t}\) is the historically observed mean proportions at age, \(\omega\) is the standard deviation in the multivariate logistic distribution of proportions at age, and \(\varepsilon_{a,t}\) are standard normal deviates (Schnute and Richards 1995). Both \(\bar{p}_{a,t}\) and \(\omega\) were estimated from historical data for each stock individually.

Beta distribution

We estimated the parameters of the beta distributions, \(\beta_1\) and \(\beta_2\), as a function of the mean, \(\mu\), and standard deviation, \(\sigma\), of a given metric (e.g., proportion of spawners that are effective females or target harvest rate), using the following approximations (as described in Morgan and Henrion 1990):
We then calculated the beta-distributed errors from $\hat{\beta}_1$ and $\hat{\beta}_2$ using the function, 

"rbeta" in R, version 2.2.

(A3) \[ \hat{\beta}_1 = \frac{\mu^2 - \mu^3 - \sigma^2 \cdot \mu}{\sigma^2} \]

(A4) \[ \hat{\beta}_2 = \frac{\mu(1-\mu)^2 - \sigma^2 (1-\mu)}{\sigma^2}. \]
Appendix 3.2 – Depensatory effect on recruitment

Depensatory mortality at low population sizes has been observed for some species, but there is large uncertainty in the magnitude of depensation (Liermann and Hilborn 1997) and it has only infrequently been documented for sockeye salmon. For example, Wood (1987) found a decline in the maximum rate of mortality of Pacific salmon during seaward migration when abundances of salmon were high due to predation by mergansers. Therefore, in a sensitivity analysis, we calculated recruitment with depensation, $R_{dep}$ in year $t$, when spawner abundances fell below a depensation threshold, $S^*$, using the following equation:

\[
(A5) \quad R_{dep,t} = R_t \cdot \left( 1 - \sqrt{\frac{S^* - S_t}{S^*}} \right),
\]

where $S_t$ is spawner abundance, and $R_t$ is recruitment calculated without depensation from equations 1 and 2, for Togiak and Chilko, respectively. We used this formulation of recruitment because when spawner abundances dropped below $S^*$, recruitment declined as a curvilinear function of spawner abundances, which is more realistic than linear declines used in some previous simulation studies such as Cass et al. (2003). We used the lowest observed spawner abundance in our empirical data as our estimate of $S^*$ (corresponding to 14.4% and 5.0% of median spawner abundance for Togiak and Chilko, respectively), because depensatory mortality has not been observed for these stocks.
References cited in Appendices


Preface to Chapter 4

In the previous three chapters I addressed uncertainties in one life history characteristic (age-at-maturity), the parameters of a forecasting model (the sibling model), and outcomes from management actions. In Chapter 3, I evaluated management procedures that accounted for those three sources of uncertainties and the interactions among them. Up until now, I have focused on uncertainties that occur primarily at local and regional scales, but the performance of management procedures will also be affected by processes affecting growth and body size at maturity that occur at a larger ocean-basin scale. For example, in some cases, Pacific salmon from different populations, countries, and species compete for a common pool of prey resources in the North Pacific Ocean, resulting in declines in growth when abundances are high (Ruggerone et al. 2003; Ruggerone and Nielsen 2004) (such as might have been caused by recent increases in numbers of hatchery-released juvenile salmon across the North Pacific). Despite potential reductions in economic value per fish commercially harvested and exacerbated conservation concerns for populations at low abundance, this decline in body size has not been addressed by international policy makers. The potential for collective action to resolve the problem of competition among salmon in the North Pacific Ocean and the best means to obtain cooperation are not clear. Thus in Chapter 4, I identify the incentives and disincentives to international collective action. By better understanding sources of behaviour of nation states, it may be possible to alter individualistic incentive structures in order to foster cooperation among countries on this problem.
Chapter 4
International Cooperation among Nation States of the North Pacific Ocean on the Problem of Competition among Salmon for a Common Pool of Prey Resources

Abstract

Nation states frequently cooperate to address common-pool resource problems such as the overexploitation of fish that straddle territorial boundaries or reside in the open ocean. However, the problem of competition among salmon populations (Oncorhynchus spp.) from different countries that compete for a common pool of prey resources in the North Pacific Ocean remains largely ignored by international policy makers. This problem is important because the resulting decrease in mean body size as abundance increases may lead to reductions in commercial, recreational, and ecosystem values. I use three prominent theoretical frameworks from the literature on regime formation in international relations, namely rational self-interest, realism, and institutionalism, to examine incentives and disincentives for cooperation among nation states in the North Pacific Ocean on this issue. I further suggest the creation of a new international organization to modify the incentive structure for collective action by using side-payments, establishing a multinational scientific assessment team to create a common frame of reference for the problem, and administering regulations, among other considerations.
Introduction

Overexploitation of fish populations that are shared among nation states is frequent in international fisheries due to common-pool resource problems (Anderson 1986). Such fisheries problems are characterized by rivalry (harvesting by one nation state hinders the ability of other nation states to harvest) and non-excludability (one nation state cannot exclude another from harvesting). In the absence of some form of collective action, such situations can result in "the tragedy of the commons", the depletion of shared resources due to incentives to overexploit them arising from the goal of maximizing individual benefit (Hardin 1968). International common-pool resources pose particular difficulties because of inter-state rivalries, concerns over relative gains among states, and lack of authoritative bodies to control the use of those resources (Burke 1997; Mitchell 1999).

For Pacific salmon fisheries (Oncorhynchus spp.), reductions in abundances have occurred for some populations due, in part, to harvest of the common-pool resource of salmon by humans (e.g., Nehlsen et al. 1991). However, salmon and harvesters are only two components of a larger ecological system, which includes a hierarchy of resource users (i.e., predators) and resources (i.e., prey). For example, although salmon are prey for human harvesters at one level, salmon also act as resource users in the open ocean where they compete with other populations and other aquatic species (salmon and non-salmon) for a common pool of prey resources (Cooney and Brodeur 1998; Aydin et al. 2003). In other words, salmon act both as a common-pool resource for human harvesters and as harvesters on their common-pool prey resources. There are interesting similarities in the relations between human harvesters and salmon populations on the one hand, and
between salmon and their prey on the other. First, reductions in the resource (salmon or salmon prey) by large numbers of harvesters (humans or salmon) may occur in both cases. For human harvesters, overexploitation is widely known to result in declining yield and in extreme cases, possible conservation concerns. For some salmon species, empirical evidence suggests that increased competition for limited prey resources can result in reductions in salmon body size due to density-dependent growth (e.g., sockeye salmon, Rogers 1980; Peterman 1984a; Pyper and Peterman 1999), and consequently reductions in reproductive output (e.g., reduced number and quality of eggs produced per female) (Forbes and Peterman 1994). These effects are important to human harvesters because smaller fish have lower economic value when harvested, and declines in reproductive output may prevent depleted populations from recovering. For example, reductions in the numbers of adults produced per spawner have been found for pink salmon (*O. gorbuscha*) populations in Prince William Sound in years when adult fish were small (covering 1975 through 1999) (Wertheimer et al. 2004).

Second, just as recreational, commercial, and First Nations harvesters compete for access to the same population of salmon, salmon of different species may also compete for prey resources. In particular, sockeye (*O. nerka*), chum (*O. keta*), and pink salmon share common prey of zooplankton and micronekton (such as squid and small fishes) (Kaeriyama et al. 2004; Aydin et al. 2003). When those species of salmon originate from different countries, it is possible for an increase in the number of salmon from one country (due to, for example, hatcheries or other types of enhancement) to contribute to increased competition for prey resources at the expense of salmon from all countries (Peterman 1984b). Furthermore, sockeye, chum, and pink salmon share the same prey
resources with large nektont (e.g., shrimp), large jellyfish, and to lesser extent, squid, marine mammals, and birds (Aydin et al. 2003), and hence competition from salmon may result in less food available for other marine species (and vice versa).

A third similarity in the relations between harvesters and salmon, and salmon and their prey, is their international context. In the North Pacific Ocean, both human harvesters and salmon originate from several countries (primarily Canada, the United States, Russia, and Japan) and prior to restrictions for fishing on the high seas that were established in 1992, the spatial distribution of harvesters overlapped on the high seas, as do salmon from different nation states where they compete for a common pool of resources. The North Pacific Anadromous Fish Commission (NPAFC) has addressed one common-pool problem, overexploitation of salmon, by banning directed fishing of salmon in the North Pacific Ocean beyond territorial limits in 1992 (deReynier 1998). In addition, various bilateral agreements (e.g., the Canada-United States Pacific Salmon Treaty) have allocated harvest of stocks that straddle territorial boundaries (Munro 2006). However, there are no analogous measures in place to deal with the next lower trophic level, i.e., to restrict the harvest of a common pool of resource prey by salmon populations from different nation states in the North Pacific (e.g., by limiting the number of juvenile fish released by each country into the ocean or by other means). Such restrictions would require considerable cooperation among nation states.

In this chapter I use three prominent theoretical frameworks in international relations to examine incentives and disincentives for cooperation among nation states in the North Pacific Ocean on this problem. The form of a cooperative agreement and hence the incentives and disincentives will depend on specific objectives. I assume two
objectives. The first is to maximize net benefits of hatchery output of juveniles into the ocean (benefits of hatchery output minus costs to hatchery and wild populations, where benefits include commercial, recreational, and ecological value of recruitment) combined across nation states of the North Pacific over the long term (100 years) (as suggested in Peterman 1991). The most significant contributors of salmon to the North Pacific are Canada, Japan, Russia, and the United States (North Pacific Anadromous Fish Commission 2005), so it is these countries that I consider in this analysis. The second objective is to share those net benefits equitably among nation states. I begin by discussing evidence for competition for a common pool of prey resources within the North Pacific among salmon populations within a species and among species and the possible effect of hatchery enhancement on that competition. International cooperation is required to address this problem because of its common-pool nature, but incentives and disincentives to cooperation have, to my knowledge, not been explored in detail. I also identify strategies and provide recommendations for international policy makers to overcome these disincentives and encourage cooperation. I limit my analysis to evaluating the potential for cooperation in reaching an agreement to achieve objectives, and do not include a full assessment of the likely effectiveness of such an agreement if it is reached.

Density-dependent growth of salmon in the North Pacific Ocean

Studies on Pacific salmon provide strong empirical evidence for density-dependent growth in the North Pacific Ocean among salmon populations within sockeye salmon (Peterman 1984a,b,c; Pyper and Peterman 1999; Ruggerone and Rogers 2003) and chum salmon (Helle and Hoffman 1998; Kaeriyama 1998)) and as well between
salmon species (e.g., between pink and sockeye salmon, and between pink and chum salmon, Ruggerone and Nielsen 2004). Specifically, those studies show that reduced age-specific body size or growth (as measured by scale circuli widths) have been observed when there is a high abundance of fish of the same or different species that share the same ocean distribution and prey. Ecological studies of prey intake from stomach contents of salmon suggest that these populations and species compete for a limited pool of temporally varying prey in relatively small and productive regions in the ocean (Tadokoro et al. 1996; Kaeriyama et al. 2004), and increased competition can reduce growth of salmon. For example, Bigler et al. (1996) found declines in mean body mass of fish caught in commercial fisheries for chum, pink, and sockeye salmon across the North Pacific from 1975 through 1993 (ranging from −2% to −32%) associated with increases in abundances. Those decreases in mean body mass occurred despite concurrent increases in mean age-at-maturity. In addition, Helle and Hoffman (1995; 1998) suggested that statistically significant declines in age-specific body length of North American chum salmon were due, in part, to density-dependent growth related to increases in abundances of chum salmon originating from both North America and Asia, which overlap in ocean distribution. Ishida et al. (1993) found similar decreases in age-specific body weight of chum salmon originating from Japan and Russia as the abundances of Japanese chum salmon increased. Furthermore, from a comparison of population abundances and scale circuli counts across species, Ruggerone et al. (2005) suggested that in years when Asian pink salmon were abundant, age-specific growth (as well as survival rates) of sockeye salmon from Bristol Bay, Alaska were reduced due to increased competition for zooplankton in the region of overlap in their ocean distribution.
Finally, bioenergetic models of prey consumption and growth of various species of salmon in the North Pacific suggest that age-specific salmon body sizes decline with relatively small decreases in prey availability (Davis et al. 1998), such as those observed with changes in environmental conditions or increased salmon density.

**Effects of hatcheries on density-dependent growth of Pacific salmon**

Enhancement of wild salmon populations with hatcheries (i.e., the release of juveniles from facilities where they are artificially reared, also known as sea ranches when located in marine water, Mahnken et al. 1983) occurs widely in both North America and Asia to supplement commercial harvest of wild populations and to compensate for declines in the quality and quantity of freshwater salmon habitat (Cooney and Brodeur 1998). Heard (1998) estimated that in 1992, about a fifth of the total number of juvenile Pacific salmon that entered the North Pacific Ocean originated from hatcheries (mostly pink and chum salmon), though this proportion was much higher for Japan and the contiguous states of the U.S. Pacific northwest. Heard (1998) also estimated that of the total 4.5 billion juvenile salmon released from hatcheries in 1992, 1.79 billion were from the United States, 1.2 billion from Japan, 871 million from Russian, and 634 million from Canada (country-specific proportions of total abundances of salmon (wild plus hatchery-origin) were not presented). Despite concurrent declines in abundance of adults in wild populations with increases in hatchery releases of chum, pink, and sockeye salmon (Mahnken et al. 1998; Wertheimer et al. 2004), some hatchery programs have been extremely successful in increasing harvest. For example, adult returns of chum salmon to Hokkaido, Japan (i.e., adults that return to freshwater and become vulnerable to the fishery) have increased by an order of magnitude from the early
1900s prior to significant development of hatcheries to the 1990s (from between 1 and 6 million adults in the period 1900 until 1960 (Thorpe 1980) to between 28 and 60 million from 1990 until 2001 (Mayama and Ishida 2003)). This increase has been attributed to a large expansion of Japanese hatcheries during the 1970s (Mayama and Ishida 2003) and a considerable increase in marine survival rate of hatchery-released juveniles due to adoption of new techniques such as fry feeding and adjustments in timing of release (Thorpe 1980). Other approaches have been used to enhance adult salmon production (i.e., numbers produced per year), such as lake fertilization and the construction of spawning channels, but hatcheries remain the dominant tool for enhancement across the North Pacific (Augerot 2001).

Although considerable research has been conducted on the effects of increased hatchery releases on the prevalence of diseases, genetic changes, and other problems in wild populations (e.g., Levin et al. 2001; Brannon et al. 2004), the mechanisms responsible for declines in body size have not been rigorously investigated. In one study of pink salmon from Prince William Sound, Cooney (1998) estimated that annual consumption of prey in coastal and oceanic feeding zones by pink salmon (wild and hatchery-origin) increased from 100,000 metric tons (mt) to over 300,000 mt after a hatchery program was introduced in 1976. He further suggested that increased competition for prey resulted in significant declines in body size of hatchery and wild pink salmon and in the number of eggs produced per spawner. Hilborn and Eggers (2000; 2001) suggested that the large observed declines in Prince William Sound wild pink salmon from 1977 to 1995 (average total abundance of returns of 16.3 million reduced to 5.2 million) were due to a combination of overexploitation by harvesters and
indirect negative impacts of hatchery fish on wild fish (e.g., density-dependent growth), among other factors. In addition, Wertheimer et al. (2004) estimated an annual loss of 1.03 million wild pink salmon (approximately 3% of total returns) in Prince William Sound due to reductions in body size from density-dependent growth and reproductive output of spawners. Moreover, Peterman (1984b) suggested that declines in body mass of some sockeye salmon populations due to density-dependent growth by up to 22% in mass for some stocks may, in fact, tend to offset the benefits of enhancement. It is possible that large hatchery releases may entirely negate benefits because of density-dependent growth, at least for some nations.

I focus my analysis on chum, pink, and sockeye salmon because concern about the negative effect of hatchery releases via density-dependent growth is greatest for those species for four reasons. (1) Those species (especially chum and pink salmon) comprise most of the hatchery output into the North Pacific Ocean (over 90% in 1992, Heard 1998). (2) The ocean distributions of sockeye salmon from populations that spawn in North America and Asia partly overlap (French et al. 1976), as do those of chum salmon (Helle and Hoffman 1995; 1998); but this is not the case for pink salmon (Ruggerone and Nielsen 2004). (3) Chum, pink, and sockeye salmon share similar diets (Aydin et al. 2003; Kaeriyama et al. 2004). (4) These species overlap in ocean distributions with each other, within and across continents of origin (Ruggerone et al. 2003; e.g., Asian pink salmon overlap with North American sockeye salmon, Fig. 4.1 here).
Potential for an international cooperative agreement

Given evidence for reductions in somatic body growth in several species of salmon across the North Pacific arising from hatchery enhancement by individual countries leading to competition for prey, there may be a need for collective action to address this common-pool resource problem. Economic benefits from fishing may have already been reduced due to large abundances of hatchery-released juveniles and the problem could get worse with further increases in juvenile abundances and/or less productive ocean conditions for salmon food. Current international agreements about Pacific salmon focus on regulating the harvest of salmon by fisheries (e.g., the 1992 Convention for the Conservation of Anadromous Stocks in the North Pacific Ocean). However, in order to reduce the effects of density-dependent growth among fish from different countries as well as within countries, it is necessary to limit the number of salmon competing for that prey. In other words, shifting the management perspective to a lower trophic level (from harvesters to salmon) (Frederick 1990) would encourage countries to focus attention on competition for the common pool of resource prey as well as overexploitation of a common pool of salmon. For example, some authors have suggested imposing quotas on salmon-producing countries to limit total combined production of juvenile salmon from wild populations and enhancement facilities, such as hatcheries (deReynier 1998; Heard 1998), or taxes on the number of adult salmon produced in each country (Peterman 1984b). Any such initiative would require a cooperative agreement among these countries (specifically, Canada, Japan, Russia, and the United States), because there is currently no international governance structure with authority to unilaterally impose measures of this nature. Although salmon fisheries also
exist in North and South Korea, total catches for those countries in 2001 were less than 1% of that for other North Pacific countries (North Pacific Anadromous Fish Commission 2005). North and South Korea are therefore not included into this analysis.

Presently, the incentives and disincentives to such a cooperative agreement on reductions in hatchery output among the relevant countries are not well understood, and these are complicated by the international context. For example, if rights to future levels of hatchery enhancement were allocated based on the historical wild production of juveniles, Japanese salmon production would suffer tremendously (deReynier 1998) because Japan’s hatchery production far exceeds its own historical wild production (Thorpe 1980; Kaeriyama 1989). This historically based allocation scheme would potentially keep Japan from becoming involved in developing a cooperative agreement. Responding to this possibility, Yasuhiro Ueno of Japan’s National Research Institute of the Far Seas Fisheries wrote:

If ... the carrying capacity is too small to carry the present salmon stocks, the latter countries [Canada, Russia, U.S. Alaskans] would probably insist one-sidedly that the former countries [Japan and U.S. mainland] should decrease the number of salmon juveniles released from their hatcheries. Such a situation is extremely adverse to Japanese national interests. It is obvious that studies on the carrying capacity for salmonids have serious political issues (cited by deReynier 1998, p.170).

To further investigate the potential for cooperation among these countries and to generate strategies to alter the incentive structure in order to encourage cooperation, in the next section, I assess the various incentives and disincentives for collective action.
Incentives and disincentives for international cooperation

The behaviour of nation states in international negotiations has been studied extensively (e.g., Krasner 1983; Young 1989; Haas 1994; Young 1994), and an analysis of this behaviour can help to identify incentives and disincentives to collective action relevant to the specific case of interactions among North Pacific salmon. The literature on state behaviour relating to the determinants for collective action offers numerous, sometimes overlapping and sometimes conflicting conclusions from various perspectives of study such as economics, political relations, and law. Examining the problem of international cooperation from more than one of these perspectives offers a deeper understanding of the potential for cooperation concerning competition for prey resources and density-dependent growth of Pacific salmon. Multi-perspective approaches have been used for analyses of other policy problems, such as international relations between China and Taiwan (Acharya 1999), sustainable development (Sunderlin 1995), risk analysis (Robinson 1992), long-term energy forecasting (Robinson 1992), and the Cuban missile crisis (Allison 1969).

Although some authors have attempted to synthesize different perspectives on international cooperation, consensus has not been reached on a complete typology of approaches for understanding state behaviour. For the present analysis, I begin with two of the most commonly employed theoretical frameworks used in international relations for understanding the formation of regimes for managing natural resources, rational self-interest, and realism (Young 1989, 1994).

Rational self-interest treats states as rational actors that seek to maximize their own utility. It compares the self-interests of different states when evaluating the potential
for cooperation, and is particularly relevant for the case of North Pacific salmon fisheries because of differences in hatchery output among countries (Mahnken et al. 1998), differing concerns over negative impacts on their own wild populations (Augerot 2001), and hence unequal costs and benefits associated with hatchery reductions.

In contrast to this utilitarian perspective, the realist framework of analysis focuses on the distribution of power (as defined by the ability of one country to impose its own preferences on others, often through control over material resources) among states (Young 1989). This realist perspective is useful for examining the North Pacific salmon fisheries problem because of the changing configuration of power that would take place as autonomy over hatchery output changes if an agreement were reached. Together, these two frameworks capture what Krasner (1983) describes as the two most important exogenous variables to explain how international governance arrangements (regimes) are created, egoist self-interest and political power (exogenous in the sense that they can cause regime formation on their own).

I supplement the utilitarian and realist frameworks by also examining the problem through another prominent theoretical framework used in the study of international relations. This framework emphasizes the role of international institutions and institutional bargaining in the development of cooperative relationships (Young 1989). In the North Pacific, there are a variety of existing international institutions that govern state behaviour and will likely influence any movement towards collective action on the problem of competition among salmon populations. Although some political scientists believe that international institutions simply reflect the underlying distribution of interests and power (i.e., they are emergent properties of rational self-interest and realism), Young
(1994, p. 30) argues that institutions are, in fact, "one of the major clusters of factors that determine the course of international affairs", and are a central explanation for collective behaviour. Similarly, Krasner (1983) includes "norms and principles" (which are embodied in institutions) as his third basic causal variable in regime development (with self-interest and political power), and other scholars also recognize institutionalism as a major school of international relations theory (Bowett 1982; Abbott 1998).

There are several related or alternative theoretical perspectives that have been used to examine and explain collective action, including liberal theory, (DeGarmo 2005), public choice theory (Young 1989), and cognitivism or social construction theory (Young 1994; DeGarmo 2005). Here, I focus only on the two main perspectives identified by Krasner (1983), rational self-interest and realism, plus institutionalism, described by Young (1989), because they capture the primary dimensions that have been used for understanding state behaviour in regime formation. They also emphasize factors that are particularly relevant to the problem of competition among salmon in the North Pacific Ocean, such as differences in self-interests among states, sovereignty and the distribution of power, and international institutions. Although these three theoretical perspectives (rational self-interest, realism, and institutionalism) have different foci, they are not mutually exclusive, i.e., the incentives and disincentives to cooperation that they identify partially overlap, as do the strategies they suggest for altering incentive structures.

In the following three sections, I describe the key incentives and disincentives for cooperation among states in the management of common-pool resources, as well as potential strategies to alter the incentive structure (simply called "strategies"), as categorized by components of the three theoretical frameworks. I also draw parallels
from various international environmental problems. To illustrate the application of the three theoretical frameworks in a marine harvesting context, I then use a case example of collective action in the management of an international common-pool resource, the North Pacific fur seal, before extending the analysis to the problem of competition for prey resources and density-dependent growth of salmon in the North Pacific Ocean.

**Rational self-interest**

Rational self-interest (or the utilitarian perspective) assumes that the interests of "players" (e.g., nation states) are known, and focuses on how their perceived costs and benefits affect their preferences for various policies (Young 1989; Young 1994). Fundamentally, each player is assumed to be a self-interested utility maximizer (Young 1989); in other words, each player chooses among policy alternatives to maximize its welfare without considering the effects on other players or its performance relative to others.

**Incentives**

Benefits to individual nation states from collective action that exceed those from unilateral behaviour create rational incentives for cooperation (Table 4.1, row 1 and columns 1 and 2). States will accept the costs associated with cooperation when it is beneficial to each of them (when a "zone of agreement" exists, Young 1994). Game theory analyzes strategic interactions among players when negotiating cooperative agreements from a utilitarian perspective, and is commonly applied to international environmental negotiations when externalities are borne across national boundaries (Finus 2001). The most frequently cited game, the prisoner's dilemma, in its simplest
form, demonstrates rational incentives for cooperation between two players with two options, cooperate or defect (i.e., do not cooperate), though it can be extended to multiple players (Munro 2006). For a single iteration of this game without communication among players, each player has an incentive to defect, despite the fact that there would be higher aggregate outcomes from mutual cooperation. This is because defecting with a cooperating player yields the highest outcome for the defecting player and avoids the losses that would be associated with cooperating against a defecting player (Finus 2001).

Cooperation becomes rational from the perspective of an individual player only when the following conditions are met: the game is repeated over many iterations, the rate at which players discount future resources is low with subsequent iterations, players communicate, and defections are monitored and punished (Garrett and Weingast 1993). According to the utilitarian perspective, under these conditions the cooperative solution is self-enforcing because it is grounded in individual rationality. The Pacific Salmon Treaty of 1985 is a good example of an institutional arrangement designed to overcome a prisoner's dilemma (Munro 2006). Without agreement on the allocation of salmon harvest, Canada and the United States faced incentives to overexploit salmon, and there were no incentives to engage in enhancement projects to increase production of adult salmon (Munro 2006). The aim of the Pacific Salmon Treaty was to prevent overfishing and provide for "optimum" production of adult salmon by regulating the allocation of harvest between states (Munro 2006).

**Disincentives**

Even when it appears that collective action would be supported by self-interested rational behaviour, cooperation is by no means guaranteed (Keohane 1994), in part
because of numerous uncertainties, including the identities of players, the nature of management options, outcomes of these options, and the order of preference for these outcomes (Young 1994) (Table 4.1, rows 2-4, columns 1 and 2). For example, in international negotiations, instead of acting like a solitary player when determining preferences, states negotiate with a large number of their own domestic agencies, which in turn reflect the interests of lobbyists and grassroots groups, thereby diversifying each state's interests and making it more difficult to determine preferences *ex ante* (Barrett 2003; DeGarmo 2005). In the case of the Pacific Salmon Treaty, actions of the United States were driven by the divergent interests of Washington and Oregon, Alaska, and treaty Native-Americans, who, in combination with Canada, created various dynamic coalitions. This resulted in uncertainties in preferences and slowed movement towards agreement (Munro et al. 1998).

Most theoretical analyses of strategic interactions assume that each player faces the same potential benefits and costs and that the resulting preferences are symmetric among players (Connolly 1999). However, inequalities often exist, potentially obstructing movement towards cooperation (Lane and Jensen 1996). Differences among states in the "shadow of the future" (discount rate) is one factor than can cause unequal values to be placed on cooperation by different states (Barkin and Shambaugh 1999). For example, when negotiating the Convention on the Conservation of Antarctic Marine Living Resources, Japan and the Soviet Union valued future harvest of krill and finfish less than the other states, giving Japan and the Soviet Union less incentive to cooperate and hence the ability to extract concessions from other states (Barkin and Shambaugh 1999).
Without effective monitoring of compliance and enforcement, some players may avoid responsibilities associated with collective action while still reaping benefits from the cooperating states. These "free-riders" externalize the costs of using common-pool resources (Connolly 1999) when calculating their utility for defection, increasing costs for cooperating states, and creating disincentives for collective action.

**Strategies**

To avoid stalemates during negotiations that occur when preferences (costs and benefits) for each player are known and determined in advance, linkages across issues can be made instead of allocating fixed resources among players (Young 1989) (Table 4.1, rows 5-7, columns 1 and 2). By linking issue areas, states explore different types of deals or outcomes outside the original negotiation space. For example, cooperation was achieved between two members of the International Whaling Commission, Iceland and the United States, but only when the controversial issue of whaling for "scientific purposes" by Iceland was linked to the United States' use of the NATO base at Keflavik in Iceland (Barrett 2003).

In addition, to foster cooperation between players with unequal values for benefits and costs, these values can be adjusted with the use of "side-payments", where countries that gain from cooperation compensate those that lose (Lane and Jensen 1996; Barrett 2003). For instance, in the 1999 Pacific Salmon Agreement, the United States created two "Endowment Funds" for Canada to appease Canadian concerns over imbalances in interceptions of each others' salmon caught by the two countries (Noakes et al. 2006). However, side payments will only be successful for rational utility-maximizing players if
all players perceive that their benefits with side-payments increase over the non-cooperative case (Barrett 2003).

Finally, the threat of "free-riders" can be diminished by ensuring that resource use is monitored, deviations are punished, and "free-riders" are excluded from receiving benefits through the use of property rights or other institutions (Stein 1983; Levy et al. 1992). If nation states perceive that the occurrence of "free-riders" is low, they are more likely to cooperate.

Realism

In contrast to rational self-interest, realism assumes that nation states are status-maximizers, not utility-maximizers (Young 1989). A state's primary objective is to gain power relative to other states in order to assure survival (Haas 1994). States evaluate their payoffs in relation to those of other states instead of in absolute terms, and are concerned with differences in gains among states instead of joint gains (Young 1989). Furthermore, states aim to preserve sovereignty, which includes the autonomy to make their own decisions, control over their resources, and the legitimacy to make rules for resource use by exercising rights recognized by other states (Liftin 1997).

Incentives

When the distribution of power is biased largely towards one state, known as a hegemon, this state can persuade structurally weak states (i.e., states with little ability to control events affecting themselves and others) into accepting terms of a cooperative agreement without overtly exercising their power (e.g., through the threat of exercising power alone) (Underal 1994; Young 1994) (Table 4.2, row 1, columns 1 and 2). Once a
powerful state gains a reputation as being an effective player in negotiations, weaker states will defer to it in anticipation of its actions (Young 1994). The theory of "hegemonic stability" claims that dominance by one powerful state is both necessary and sufficient to achieve cooperation, though the end result may not be collectively desirable (Barrett 2003). Instead, the resulting agreements tend to align with the interests of the hegemon (Haas 1994). Even when the bias in power is less extreme (i.e., without an obvious hegemon), according to realism, the collective agreement will be shaped by the most powerful state (Haas 1994). For example, the United States was able to pressure Japan into an agreement on banning drift nets in the North Pacific Ocean by threatening restrictions on the import of Japanese fish products (Young 1994). Because the United States is a more important market for Japanese products than the reverse, the threat of import restrictions was enough to attain Japanese compliance on banning drift nets.

**Disincentives**

Empirical evidence suggests that hegemons, however, are not always successful in imposing agreement on other states. For instance, weaker states may not cooperate because they fear further losses in their own power and sovereignty (Barrett 2003) (Table 4.2, rows 2 and 3, columns 1 and 2). One case study involving Baltic Sea states suggested that negotiations towards cooperative agreements on environmental issues may falter if economically poor post-Communist states fear excessive loss of power and prestige through "bullying" from rich capitalist states (Lane and Jensen 1996; Barrett 2003). In contrast to "hegemonic stability theory", maximizing status is seen by some realists as a zero-sum game (i.e., if one state gains power, then other states necessarily lose power). Therefore, in such situations, states are in constant conflict with little hope of reaching
agreement to realize joint gains (Young 1989). Furthermore, cooperative agreements
often entail a ceding of sovereignty as defined by its three components: autonomy or
independence in decision-making and action; control or the ability to produce an effect;
and legitimacy or the recognized right to make rules. In contrast, states driven by power
interests insist on maintaining sovereignty over decision-making and resources (Liftin
1997). In fact, Roberts (1996) suggests that a fear of loss of sovereignty is a central
factor limiting cooperation among states. For example, states such as Brazil and
Malaysia were strongly opposed to provisions of the UN Biodiversity Convention that
curtailed their authority over logging practices in their own jurisdiction (Barrett 2003).

**Strategies**

If threats by states with the greatest power are not sufficient to reach agreement,
these states can use their material resources to induce other states to agree by offering
rewards or imposing punishments (Young 1994) (Table 4.2, rows 4 and 5, columns 1 and
2). As for the utilitarian perspective, this transfer of resources redistributes the cost and
benefits of the transaction, potentially making cooperation more attractive to some
players. However, according to realism, such transfers originate from an imbalance of
power among states and may influence the subsequent power hierarchy among states
(e.g., a state may gain the reputation of being a "strong" or "weak" negotiator by giving
or accepting transfers) (Mäler 1991; Young 1994).

Furthermore, to avoid losses in total sovereignty, states may trade off different
components (autonomy, control, and legitimacy) in "sovereignty bargains" (Liftin 1997).
For example, a state may sacrifice some autonomy in international agreements, in order
to increase their ability to control the resource under negotiation, or to be recognized on
the international arena as having the legitimate right to create rules for resource use (Liftin 1997). For instance, Brazil and Malaysia may accept an initial loss in autonomy over decision making on logging practices in their territories for increased legitimacy on the world stage as an active player when negotiating international agreements on biodiversity (Barrett 2003).

**Institutionalism**

According to institutionalism, institutions can resolve one of the greatest barriers to cooperation, "free-riding", by providing rules for behaviour and by assuring that states that deviate from these rules will be detected and punished (Stein 1983). Institutionalism, then, examines the institutional context or rules under which cooperation will be pursued by self-interested states (Haas 1994). In this way, institutions are not merely the outcome of successful negotiations among states, they also facilitate the creation and maintenance of international cooperation (Levy et al. 1992). Institutions can take the form of international organizations, or less formal rules or norms for behaviours (Liftin 1997).

**Incentives**

Assuming that states are rational actors, they can be expected to voluntarily create institutions as long as the likelihood of repeated interactions is high (i.e., a state's choice of a strategy for addressing a problem is not a one-time event) and transaction costs (costs of negotiating and enforcing an agreement) are low (Lane and Jensen 1996). In other words, given these conditions, small groups of states will establish institutions or rules that guarantee the highest possible mutual gains, also known as the "first-best" solution (Lane and Jensen 1996). International institutions foster cooperation in several
ways: by increasing communication among states, linking apparently disparate issues, encouraging reciprocity, increasing information flow among states (and hence reducing inequalities resulting from differences in research capabilities), monitoring state behaviour, and reducing the number of strategic moves among states by creating rules for behaviour (DeGarmo 2005) (Table 4.3, row 1, top of columns 1 and 2). Furthermore, DeGarmo (2005) suggested that the structure of the international system at the time of negotiation (for example, the presence of over-arching environmental agreements, such as the FAO Code of Conduct for Responsible Fisheries (Food and Agriculture Organization 1995) for the case of marine fisheries) is an important determinant of participation in cooperative environmental agreements.

**Disincentives**

Despite assertions from the utilitarian perspective that rational nation states will choose the cooperative agreement and corresponding institutions that maximize their respective mutual gains, this "first-best" solution may be either infeasible or undesirable (Lane and Jensen 1996) (Table 4.3, rows 2 and 3, columns 1 and 2). Cooperation will be infeasible if a state lacks the economic resources, information, or political support to participate in institutions (Lane and Jensen 1996), and undesirable if substantial differences in preferences (Lane and Jensen 1996) or power (Young 1989) exist among states. For example, institutions created to ameliorate pollution in the Baltic Sea may be infeasible for economically poor, former Communist states (e.g., Lithuania and Latvia) that do not have either the resources to implement them or a history of domestic public support for environmental issues (Lane and Jensen 1996). In addition, institutions
imposed by a hegemon may not be in the interest of weaker states and therefore may be undesirable (Young 1989).

Failures to reach feasible joint gains are common in international fisheries negotiations, in part because of uncertainties in preferences and in predictions of outcomes of various institutional arrangements (Young 1989). These uncertainties may hinder cooperative agreements in several ways. First, there may be a large number of stable paths to cooperation with no easily apparent way for a set of decentralized states to converge on a single path (Garrett and Weingast 1993). Second, incomplete information about outcomes may result in unforeseen contingencies that agreements are unable to address (Garrett and Weingast 1993). Third, bargaining may require simultaneous and difficult trade-offs among equity, ecological integrity, and allocative efficiency (i.e., the allocation of resources in which it is impossible to make any player better off without making at least one player less well off), among other factors (Young 1989). In environmental negotiations, these failures have been most prevalent in issues plagued with uncertainties and conflicting values such as efforts to address global climate change, deep sea bed mining (Young 1994), and trade in endangered species (Young 1989).

Strategies

Instead of arising solely from the operation of individual rational self-interest or power assertions (i.e., by a hegemon), institutions can be created by social or institutional bargaining (Young 1989) (Table 4.3, row 4, columns 1 and 2). From utilitarian and realist perspectives, self-conscious efforts to design institutions are unnecessary for two reasons. First, according to the utilitarian perspective, the preferences of states are known and institutions that will accrue the highest net-benefits can be predicted a priori (Young
Second, according to the realist framework, the dominant state can pressure other states to accept the arrangements that it prefers (as in the case of the United States pressuring Japan to enter into an agreement to ban drift-net fisheries in the North Pacific Ocean) (Young 1994). In contrast, in institutional bargaining, the creation of institutions is not a spontaneous outcome of rational cost-benefit calculations or one-sided imposition (Young 1989), but is influenced by social factors such as leadership, perceptions of equity, authority (e.g., the authority of coastal states to manage fisheries in coastal water) and definition of roles (e.g., the role of port states instead of flag states to deal with pollution from vessels) (Young 1999).

For institutional bargaining to be successful, Young (1989, 1994) and others suggest several conditions that must be met during negotiation, some of which overlap with strategies from the other theoretical perspectives previously mentioned. First, the instruments and outcomes must be perceived as fair by all parties in order to gain political appeal (such as in the agreement for equal percentage cuts in CFCs across industrial states to address ozone depletion in the 1987 Montreal Protocol) (Young 1994; Endres and Ohl 2002). In some cases, a fair distribution of resources may override demands for economic efficiency (i.e., parties may be willing to choose a less-efficient policy option to ensure a fair distribution of resources) (Endres and Ohl 2002). Second, for some common-pool resources that attract public interest, to ensure domestic political feasibility the views of sub-national, scientific, and citizen groups may need to be accounted for during negotiation (Young 1994; Endres and Ohl 2002; Uitto and Duda 2002). For example, an analysis of three case studies of international negotiations on water use plans in Africa, Asian, and Latin America found that involvement by sub-
national groups (governmental and non-governmental) were critically important to achieve peaceful cooperation (Uitto and Duda 2002). Third, to avoid stalemates that occur when preferences cannot be determined in advance (i.e., when zones of agreement are not clear), institutional bargaining can be used to expand the negotiation space beyond that identified in traditional distributive bargaining (i.e., bargaining involving the distribution of known costs and benefits). A larger negotiation space may change motives for each state to participate (Young 1989). Fourth, scientific learning and new consensual knowledge can resolve disputes originating from uncertain consequences of policies, and increase the negotiation space for agreement by linking apparently disparate issues (Haas 1994). On the other hand, scientific uncertainty can be beneficial to the extent that it blurs positions in the zone of agreement. When uncertainties are large, the distributions of costs and benefits cannot be identified exactly, and it may be unclear who will be in the position of winner or loser. This uncertainty may promote the common good instead of individual self-interest if the preferences for costs and benefits are similar among players (Young 1989). For example, when negotiating agreement on the problem of transboundary nuclear fallout from nuclear accidents, uncertainties about what role each state will find itself in (e.g., site of the accident, victim, or bystander) are helpful because each is then more likely to consider the mutual gains instead of one-sided self-interest (Young 1994). Fifth, simple, clear focal points should be identified to facilitate communication among sub-national and national groups (Young 1994), help negotiations among states converge to a single cooperative path (Garrett and Weingast 1993), and guide behaviour around unforeseen contingencies (Garrett and Weingast 1993). For example, equal percentage cuts in CFCs across the board provided a focal point during
negotiations on ozone depletion (Young 1994). Finally, compliance mechanisms should be transparent (simple to verify) and easy to enforce (requiring little political effort and few resources) (Young 1994). If these conditions are met, institutional bargaining can promote cooperation among states.

Fur seal case study

International management of the North Pacific Fur seal provides a clear illustration of the application of these three theoretical frameworks in a marine harvesting context. Here I outline the incentives and disincentives for cooperation, and strategies to overcome disincentives for this issue (see Table 4.1-4.3, columns 1-3 for more detailed descriptions), before extending my analysis to competition among salmon in the North Pacific Ocean.

The North Pacific Fur Seal Treaty of 1911 has been described as an "unqualified success" in cooperatively resolving overexploitation of a common-pool resource (Barrett 2003). Fur seals were first hunted by Europeans in the 1700s for their luxurious pelts. Although Russia and the United States had exclusive rights to harvest seals on their terrestrial breeding grounds in the North Pacific Ocean in the late 1800s, pelagic sealing (i.e., sealing on the high seas beyond the historical three-mile territorial limit) was open to all nations. However, sealing on land was much more efficient than pelagic sealing because seals could be herded and easily slaughtered on land, whereas in the ocean most speared seals (approximately four out of five) escaped capture and eventually died directly from injuries or indirectly from predation. While states with territorial rights to breeding grounds tried to protect the species (for example, by nationalizing the harvest and regulating the magnitude of the harvest at purportedly sustainable levels), states
involved in pelagic sealing (Canada and Japan) did not restrict harvesting effort, thereby threatening the survival of the species (Dorsey 1991; Barrett 2003). Despite efforts by Russia and the United States to prevent pelagic sealing through military force, diplomacy, moral leadership, and arbitration, dramatic declines in fur seal abundances from overexploitation continued into the early 1900s (from a population of 2 million at its peak in the 1700-1800s to 150,000 in 1909) (Barrett 2003). By 1911, declines in economic rent (profits above normal levels) associated with reduced pelagic catches shifted the priorities of Japan and Canada towards conservation, prompting the signing of the North Pacific Fur Seal treaty by the United States, Russia, Japan, and Great Britain (on behalf of Canada). The success of the treaty was evident in the recovery of the fur seal population (by 1940 the population size was again greater than 2 million) and the stability of the agreement, which survived until 1940 when Japan withdrew due to increasing hostility with the United States during the Second World War. The treaty was revived again in a revised form in 1957 (Kasahara and Burke 1973).

According to the utilitarian perspective (Table 4.1, column 3), although interests, alternatives, and outcomes were clearly defined in the fur seal problem, the ratios of costs to benefits were unequal among states, in part because of differences in the implied discount rate applied to future harvests. In particular, when estimating the value of the harvest, Russia and the United States valued future harvest more (i.e., they appeared to use a lower discount rate) than Japan and Canada because the benefits accruing to Russia and the United States were primarily from harvesting seals on the breeding grounds, which, when harvested at a low rate and in the absence of pelagic sealing, gave large yields over the long term. In contrast, in the absence of regulations or property rights over
pelagic sealing, the benefits accrued to Japan and Canada were primarily from participating in a "race to fish", which implied a high discount rate on the value of future harvest. During negotiations, side-payments were used to overcome this potential disincentive to cooperation by allowing states that gained from a ban on pelagic sealing (the United States and Russia) to financially compensate those that lost from that ban (Japan and Canada). Furthermore, the 1911 treaty created an institution to monitor compliance and set requirements for domestic enforcement in order to prevent "free-riders".

From the realist perspective (Table 4.2, column 3), although no hegemon was present in the North Pacific to impose an agreement, there was a perceived asymmetry in power at least initially between Japan and Russia that created a disincentive for cooperation during earlier negotiation. Russia believed that its claims over fur seals carried more weight because of its longer history of sealing than Japan and greater structural power (i.e., the ability to control material resources, Young 1994). However, after Japan's victory in the Russo-Japanese war of 1904-1905 and acquisition of Robben Island in 1906 with its own fur seal population, Japan began asserting its rights over pelagic and terrestrial harvest of fur seals (Dorsey 1991; Barrett 2003). In addition, the perceived loss of power by Japan and Canada from foregoing pelagic sealing was compensated for by side-payments from Russia and the United States. Furthermore, using "sovereignty bargains", the loss of autonomy by Japan and Canada over the ability to harvest seals on the high seas was offset by an increase in international legitimacy by participating in the agreement and an increase in control over the resource by using cooperative action to prevent further declines in abundance (Young 1981).
From an institutionalist perspective (Table 4.3, columns 1-3), previous bilateral agreements set the stage for the 1911 treaty (Barrett 2003) by increasing communication between states, focusing negotiations on one key strategy (a ban on pelagic sealing), and establishing a precedent for side payments in exchange for such a ban. For example, in 1908, Russia sought an agreement with Japan banning pelagic harvesting in the western Pacific, and in February of 1911, the United States and Great Britain (on behalf of Canada) signed a bilateral agreement giving Great Britain compensation for a ban on pelagic sealing.

In addition, the 1911 treaty created joint property rights by designating fur seals the common property of all states that were party to the treaty, but giving immediate managerial responsibility to the United States and Russia (Young 1981) (Table 4.3, column 3). For example, input from Canada and Japan on important management decisions was deemed necessary when the United States decided to institute a moratorium on commercial harvesting of fur seals on Saint George Island in 1973 (Young 1981), reflecting the property rights of Canada and Japan over the resource. Furthermore, the 1911 treaty facilitated cooperation by incorporating effective deterrents on non-compliance and non-participation. In particular, states with breeding populations were obliged to monitor the high seas for pelagic sealing, and vessels that violated the ban were punished in their state of origin (Barrett 2003). Non-participation in the agreement was discouraged by banning imports of seal skins from states not party to the treaty, a relatively easy task since all seal skins had to be processed in London, the only port city with the necessary technological expertise (Barrett 2003).
Pacific salmon

Although several authors (Mahnken et al. 1983; Peterman 1984b; Heard 1998) have suggested reducing international hatchery output to resolve the problem of competition for prey resources and density-dependent growth of salmon (especially for pink, chum, and sockeye salmon), no such initiatives have been taken. The fourth column of Tables 4.1, 4.2, and 4.3 lists incentives and disincentives for collective action, and possible strategies for altering the incentive structure to increase the likelihood of cooperation as described below in more detail.

Rational self-interest

Under the utilitarian framework, the benefits associated with a reduction in hatchery output of juveniles from all countries and the resulting increase in age-specific body size of salmon may exceed the costs of foregone catch of hatchery fish, though the distribution of these benefits and costs will differ among countries (Table 4.1, row 1, last column). Benefits are derived directly from increased biomass per fish of commercial catches of wild populations (i.e., populations that spawn naturally) together with increased biomass per fish of catches of hatchery-origin fish. For example, for the Fraser River, Canada, an increase in body length equal in magnitude to the average decline in length observed for this population from 1948-1994 (Pyper et al. 1999) (which includes both wild and hatchery-origin sockeye salmon) corresponds to an increase in value of $960,875 CDN to the commercial fishery when applied to the 2003 fishery (in 2003 dollars), equivalent to 9.6% of the total value of the catch in that year (see Appendix 4.1 for calculations). In addition, the recreational fishery on wild populations may become more valuable if the abundance of "trophy" fish (i.e., large fish conferring high social
status to anglers and economic value to the recreational fishery, Pope et al. 2005) increases with average body size, but this relation has not been quantified. Although the proportion of hatchery fish that are "trophy" size may also increase as body size increases, the overall abundance of hatchery-origin fish will decline with reductions in hatchery output of juveniles, so the net benefits depend on the relative magnitude of these two factors.

Benefits of increased body size may also be derived indirectly from changes in population dynamics and the provision of ecosystem services. Specifically, increases in body size of wild salmon may result in higher reproductive output (i.e., more and higher quality eggs) (Forbes and Peterman 1994; Cooney and Brodeur 1998) and potentially larger population sizes (Wertheimer et al. 2004). However, these increases in population size would likely be several orders of magnitude less than those associated with augmented hatchery releases of juveniles observed during the 1970s (Mahnken et al. 1998; Wertheimer et al. 2004). In addition, at large population sizes, density-dependent survival rates (i.e., lower survival rates at high abundance, Walters and Martell 2004) may dampen further increases in abundance with increased reproductive output. Nevertheless, potential increases in abundance of wild populations may in turn enhance commercial, recreational, and ecosystem values of these populations. For example, some freshwater ecosystems rely on inputs of nutrients from salmon carcasses, so increased abundances of salmon may augment productivity in nutrient-limited lakes where wild populations spawn (Gende et al. 2002). Furthermore, increases in body size and reproductive output may facilitate recovery of depleted populations. Although hatchery-origin fish may also be larger and have a higher reproductive output with reductions in hatchery releases and
competition among salmon, declines in catch of hatchery-origin fish will negate any potential increases in population size.

The most important costs of foregone hatchery output of juveniles will be reduced commercial and recreational value of hatchery-origin fish. In contrast to wild populations that will not have lower abundance, reduced hatchery-origin populations may lead to declines in those values derived from commercial and recreational fisheries, and ecosystem processes, despite potential increases in value per fish from larger body sizes.

Therefore, the total benefits and costs of reductions in hatchery output of juveniles for individual populations differ according to origin (wild or hatchery) and status (abundant or depleted) and the proportion of each of these types of populations will vary among countries and regions within countries. For example, in 1995, adult salmon of hatchery-origin comprised approximately 95% of total harvest in Japan, but only 6% in the Alaska peninsula and Kodiak Island. In addition, the United States has more than triple the number of salmon populations that are listed as endangered compared with Japan, Canada, or Russia (State of the Salmon Consortium 2006). From a synthesis of biological reports from Japan, Russia, Canada, and the United States, Augerot (2005) determined that the Pacific northwest of the United States had the highest concentrations of high-risk wild populations (i.e., populations with fewer than 200 individuals over the most recent five years, have experienced declines, and return fewer than one recruit per spawner) of all regions in the North Pacific. One caveat of this analysis was that stock-specific data on wild populations from Japan were of poor quality because the Japanese fishery is dominated by hatchery-origin fish (Augerot 2005).
From the utilitarian perspective, while positive net benefits (benefits minus costs) of a reduction in hatchery output may create incentives for collective action for some countries, three disincentives exist (Table 4.1, rows 2-4, column 4). First, although the identity and interests of players are known, the magnitude and distribution of the effects of reducing hatchery output of juveniles on body size of salmon from domestic and international sources remain uncertain (Table 4.1, column 4). In addition, the extent that growth of salmon from a specific country is affected by competition for prey resources from its own salmon compared to the effects of competition from salmon of other countries is unknown. Furthermore, the magnitude of competition for prey resources from populations of the same species compared to competition from different salmon species has also not been quantified, nor has the difference in competitive ability of salmon originating from various countries, due to, for example, differences in body size during ocean life-stages.

Second, costs and benefits that differ among countries may deter cooperation. If reductions in hatchery output of juveniles per year are based on historical wild production (i.e., number of juveniles produced from wild populations per year), the largest costs will be borne by those countries with the largest difference between current hatchery output and historical wild production (deReynier 1998). For example, costs will be large for Japan, where total abundance of adult returns of chum and pink salmon have increased by an order of magnitude since the 1960s largely due to increases in hatchery output and improved technology (Kaeriyama 1999). These costs most likely would exceed any benefits to Japan from an increase in body size. In contrast, benefits associated with larger body sizes and increased reproductive output may be greater for countries that
have the greatest number of depleted (but extant) wild populations. However, because of a lack of systematic monitoring and reporting in some countries (e.g., Russia and Japan), the status of many wild populations is uncertain, so these potential benefits cannot be compared quantitatively among countries.

Third, although non-compliance with any requirements to decrease hatchery output could potentially be detected by the NPAFC's current data collection program (which includes hatchery releases), two disincentives exist to using such data to monitor a cooperative agreement on hatchery production. The NPAFC is the international organization created from the Convention on the Conservation of Anadromous Fish Commission (1993) to monitor and enforce a ban on directed fishing of salmon in the North Pacific Ocean, and currently collects data on hatchery output reported by member countries (including Canada, Japan, Russia, and the United States). The first disincentive is the delay between occurrence of hatchery output and compiling data in NPAFC Statistical Yearbooks (typically about 5 years) due to logistical and bureaucratic impediments, which prevents these data from being a timely means of detecting excess releases. Second, there are no systematic procedures in place to monitor the quality of these data. In addition, no institutional structure exists to regulate hatchery releases, further reducing the incentive for collective action.

From a utilitarian perspective, several approaches can be used to alter the incentive structure to promote collective action, including creating institutions, linking separate issues, and using side-payments (Table 4.1, rows 5-7, column 4). A new international institution including Canada, Japan, Russia and the United States could be created to regulate total hatchery output, monitor compliance, and punish cheaters
There are at least two international North Pacific fisheries organizations relevant to this topic. Although the NPAFC promotes a precautionary approach to managing aquatic species of the North Pacific Ocean by enforcing a ban on directed high seas salmon fishing (Burke 1994; deReynier 1998) and engaging in research on ecologically related species, its founding convention focuses on restricting overexploitation by harvesters on a common pool of salmon. The shift in management perspective to a lower trophic level necessitates a different approach to management than those approaches advocated by the NPAFC (such as restrictions on hatchery releases instead of on salmon harvest). The other international organization in the North Pacific, the North Pacific Marine Science Organization, PICES, focuses on improving scientific understanding of the North Pacific ecosystems, including interactions among species, but has no managerial authority. Therefore neither organization currently has the capacity to explicitly address competition by salmon for the common pool of prey resources in the ocean. A change in the mandate of at least one of these organizations would be necessary to address this problem.

For problems involving salmon management in the North Pacific Ocean, the threat of "free-riding" by countries other than those members of the institution is low because, with the exception of North and South Korea, whose total catches are small compared with those of Canada, Japan, Russia, and the United States, no other countries have populations of salmon that migrate to the North Pacific Ocean (deReynier 1998). Although issue linkages are commonly used in institutional bargaining to enlarge the negotiation space, policies that link hatchery output to other issues have not, to my knowledge, been explored by policy makers of the North Pacific and could be useful if a
stalemate occurs. For example, reductions in hatchery releases could be linked to harvest of other fish species or mammals in the North Pacific (e.g., North Pacific fur seal) or to allowances for the commercial harvest of whales, which are currently limited to being taken for scientific purposes only under the International Whaling Convention (International Whaling Commission 2006). Finally, side-payments could be used to compensate countries that would otherwise lose from an agreement that reduced hatchery releases. These side-payments could take the form of financial transfers from those countries that would gain to those that would lose, research and technology into methods to reduce competition among populations from different countries, or funds directed to conservation of depleted wild stocks.

Realism

From the realist perspective, although no clear hegemon is currently present in the North Pacific (Kaye 2001) and the balance of power with respect to fisheries management is relatively symmetric (deReynier 1998), changes in sovereignty can be expected from a cooperative agreement (Table 4.2, rows 1-3, column 4). However, as demonstrated in the fur seal example, losses in absolute sovereignty by any one country can be avoided by exploiting "sovereignty bargains" (Table 4.2, row 5, column 4). For example, a loss of autonomy over domestic enhancement facilities (i.e., ability of a state to make decisions about the number of fish released from its own hatcheries) may be acceptable if it coincides with the ability of that state to influence hatchery releases in other countries (through, for example, an international agreement), and hence increased control over the change in body size of harvested salmon (i.e., ability to constrain density-dependent growth of salmon and its consequences). This may be especially
important for the United States, which has large numbers of depleted populations (State of the Salmon Consortium 2006) and strong domestic support for recovery of these populations (as reflected in legislation such as the Endangered Species Act). That is, the United States might be willing to give up autonomy over setting the number of releases from its own hatcheries in return for seeing declines in releases from other countries and the benefits to size of adults of US-origin salmon. Alternatively, increased control over changes in body size and recovery of depleted species may create little, if any incentives for cooperative action by either Japan (since their harvests consist mostly of hatchery-origin chum salmon), or Russia (since they have a history of weak domestic support for ecological conservation) (Augerot 2001).

**Institutionalism**

From an institutionalist perspective, previous international agreements that address key management and ecological principles, such as the precautionary approach to fisheries management and ecosystem-based management (Kaye 2001; Tsuru 2004), may encourage countries both to move towards collective action and provide templates for that action. In fact, the precautionary approach and accounting for ecosystem effects (such as interactions among species originating from different countries) may already be emerging as norms of customary international law (Kaye 2001).

For example, the 1995 UN Agreement on Straddling Fish Stocks and Highly Migratory Fish Stocks (herein referred to as the Straddling Stocks Agreement), signed by 78 countries including Canada, Japan, Russia and the United States, states that, "the absence of adequate scientific information shall not be used as a reason for postponing or failing to take conservation and management measures" (Tsuru 2004). When applied to
salmon in the North Pacific Ocean, this concept could be used to support management actions designed to stop declines in body size before resolving remaining scientific uncertainties. In addition, the Convention for the Conservation of Anadromous Stocks in the North Pacific Ocean (1992) promotes "progress of science on the productivity and conservation of anadromous stocks and the conservation of ecologically related species" (North Pacific Anadromous Fish Commission 1993). In fact, the issue of competition for prey resources and density-dependent growth of salmon resulting from large increases in hatchery output is a major concern of the NPAFC (deReynier 1998). At present, however, this concern is reflected only in a mandate for scientific research, with no detailed consideration of how to implement corrective measures to deal with ecological interactions among species.

Furthermore, Principle 2 of the 1992 Rio Declaration on Environment and Development, signed by 178 governments, proclaims that each country has the "responsibility to ensure that activities within their jurisdiction or control do not cause damage to the environment of other states of areas beyond the limits of national jurisdiction" (Freestone 1996), a concept that is becoming accepted as customary law (Muldoon 1993). This view implies that any hatchery operations by one country should not harm salmon populations from another country through, for example, density-dependent interactions in the North Pacific Ocean.

In addition to the role of previous international agreements, the creation of a new institution may create further incentives for collective action by preventing member countries from "free-riding" (Joyner 1975; Frederick 1990; Heard 1998). Countries may be more willing to reduce hatchery output if regulations, monitoring, and enforcement are
formalized in an institution because each country can then be relatively confident that cheaters will not free ride on their efforts.

Although a new institution, or a new mandate for an existing institution such as the NPAFC, may encourage compliance and promote trust among countries, it may not overcome the domestic political difficulties associated with a removal of enhancement facilities or reduction in hatchery output (Table 4.3, column 4). For example, despite scientific concerns over the ability of hatcheries to achieve harvest and conservation goals (Hilborn and Eggers 2000; Levin et al. 2001), recent reductions in government funding for British Columbia, Canada's "Salmon Enhancement Program", which would result in reduction of hatchery output, have been met with strong public criticism from various environmental NGOs and recreational fishing groups (e.g., The Province 2002; The Vancouver Sun 2004) because of possible declines in abundance of populations that are currently enhanced by hatchery output. However, at least among the international scientific community, the role of hatchery releases of juveniles on density-dependent marine growth of salmon is emerging as a common concern (including among scientists from Japan, the country with potentially the most to lose from an agreement restricting hatchery releases) (Mahnken et al. 1998; Kaeriyama 1999).

Uncertainty concerning property rights to prey resources for salmon in the North Pacific Ocean may further impede movement towards collective action. While previous international agreements have regulated commercial fisheries on the high seas, they seldom assigned property rights. The UN Convention on the Law of the Sea, UNCLOS (1958), maintained freedom of fishing outside the EEZ of all countries. Although subsequent amendments to UNCLOS in 1982 obliged countries to cooperate for
conservation purposes, this did not rescind freedom of fishing on the high seas (Shram and Tahindro 1999). One agreement that has allocated rights to harvest fish on the high seas in efforts to protect depleted populations is the Southern Bluefin Tuna Convention, signed by Australia, New Zealand, and Japan (Berguin and Haward 1994). For this highly migratory species, a large portion of the most productive fishing areas lie within those nations' Exclusive Economic Zones (EEZs), limiting impacts of free-riders (i.e., nations other than Australia, New Zealand and Japan) on conservation efforts of member countries. Other agreements such as the Straddling Stocks Agreement have strengthened the rights of regional fisheries organizations to limit fishing on the high seas and within EEZs of coastal states for conservation purposes (Burke 1994). However, there are no provisions for fisheries management actions of coastal states to be compatible with that of other coastal states (e.g., for hatchery output of one state to be compatible with conservation efforts of another state). For the case of Pacific salmon, no international agreements have explicitly addressed property rights over prey species of the harvestable fish.

Shared depletable resources have been successfully allocated among countries for other natural resources such as transboundary oil, and to a limited extent, groundwater reserves, by comparing utilitarian costs and benefits (as described by rational self-interest), and using institutional bargaining (by focusing, for example, on equity among countries, domestic feasibility, and simple, clear focal points during negotiations) (Young 1994; Endres and Ohl 2002). These natural resources have some similar characteristics to prey resources used by salmon in the North Pacific Ocean, since their distribution is mobile (resources move through the available space or habitat), and they are shared and
depletable. However, unlike oil and groundwater reserves, prey resources used by salmon are accessible to all salmon (i.e., they do not lie within any one country's territory). Despite this difference, management approaches may be transferable across environmental issues. Oil reserves are typically managed cooperatively by either first assigning property rights according to the proportion of the reserve that lies within each state's territory and then allocating costs and benefits accordingly, or using another agreed-upon compromise (Eckstein and Eckstein 2005). Often, one state (or company) is charged with extracting the oil, and then allocates the costs and benefits to the other states (or companies). Several authors have suggested that this cooperative approach to management could be applied to groundwater reserves (Caponera 1992; Eckstein and Eckstein 2005), and it may also be suitable for salmon prey in the North Pacific Ocean (e.g., by allocating harvest of prey resources by salmon from different countries based the magnitude of historical and current use of this resource and notions of equity). One challenge when attempting to allocate natural resources, however, is combining the economic costs and benefits of resource extraction with the perceived intrinsic rights to resources (Eckstein and Eckstein 2005). For example, ground water is often considered a basic human necessity, and some argue that its value cannot be quantified with conventional economic analyses (Great Lakes Fisheries Commission 1997). A similar argument could be made for the intrinsic right for nation states to access the common pool of prey resources in the North Pacific Ocean via their salmon. To address the challenge for transboundary ground water within countries, permit systems have been employed, where allowable extractions of water or permits are negotiated based on a combination of equitable distribution and economic efficiency (Caponera 1992).
However, this approach has yet to be applied internationally (Caponera 1992; Frisvold and Caswell 2000).

A further complexity to allocating access to resources is diversity among resource users. For example, ground water can be extracted for agricultural use or supplied to lakes and rivers as part of a larger hydrologic system (Caponera 1992). Similarly, prey resources in the North Pacific Ocean can be used by salmon as well as other aquatic species (e.g., squid, seabirds, and pelagic forage fish, Aydin et al. 2003). Exploitation by one user (salmon or other aquatic species) may infringe on the exploitation by another, but the magnitude of the interactions between competing users is uncertain. For ground water, international institutions, such as the Commission on Transboundary Aquifer Resource Management (CTARM), formed in 1997, have made progress towards resolving these scientific uncertainties, raising awareness about the nature of the allocation problem, and promoting communication among users and states (Puri and Aureli 2005). The institutional approaches used for groundwater have been applied to other open access resources (e.g., rights to emit CO₂ allocated according to a permit system established by the Kyoto Protocol (1998)), and may also be applied to the problem of competition of Pacific salmon for prey.

Recommendations

To facilitate collective action among nation states to deal with international implications of any one country's measures to increase abundance of its salmon populations, I recommend creating a new institution (as demonstrated above from utilitarian and institutionalist perspectives) in the form of an international organization. Its mandate would be to maximize the net benefits of hatchery and wild output (benefits
minus costs to hatchery-origin and wild populations) combined over nation states of the North Pacific (Canada, Japan, Russia, and the United States) over the long term while ensuring the equitable distribution of those net benefits and recognizing the broader ecosystem impacts of competition for prey resources with other marine species. This mandate could be partially (but not completely) achieved by collaborating with other international organizations in the North Pacific that have related goals, so the new organization could be relatively small in size. Specifically, a new organization could contribute to collective action in four ways (as summarized in Table 4.4).

First, such an organization could help resolve key uncertainties and create a common frame of reference among countries for addressing the problem. Multi-national scientific assessment teams could be established (Levy et al. 1992) to work collaboratively with other international organizations (e.g., the division of PICES that investigates Climate Change and Carrying Capacity, named CCCC) to assess uncertainties pertaining to salmon body size and the effects of reductions in hatchery releases on competition for prey resources by salmon in the North Pacific Ocean. In addition, research on the economic valuation of prey resources and approaches to limiting hatchery releases by states on an equitable and economically efficient basis would help create a common framework for negotiations among nation states. Funding for these assessment teams could be supplied by member states, where financial contributions by individual countries could be based on the number of juveniles their hatcheries release.

Second, such an organization could administer rules governing the exploitation of prey resources in the North Pacific Ocean by regulating hatchery releases. The choice of regulations will depend in part on valuation of costs and benefits by country. Options for
regulations include quotas restricting hatchery output by equal proportions across countries or to historical abundances of juveniles produced per year from wild populations (Joyner 1975), taxes on each fish released (Peterman 1984b), or tradeable permits, i.e., a transferable right to release juveniles from hatcheries (Ellerman 2005). Combinations of such regulations are also possible.

Diversity in magnitude and sources of value of salmon among countries may result in diversity in preferences for alternative policies. For example, Japan may prefer equal proportional reductions in hatchery releases or an equal tax per fish released across countries instead of reductions to historical levels of wild production because annual hatchery outputs of juveniles from Japan have increased far beyond historical wild production of juveniles per year. On the other hand, Canada and the United States may prefer reductions in hatchery releases to historical levels (prior to large hatchery expansions in the 1970s) or tradeable permits assigned initially according to historical wild abundances to avoid any reductions in their output from hatcheries that are used to enhance depleted wild stocks. As identified from the utilitarian framework, side-payments could also be administered by the organization to compensate Japan for reductions in hatchery releases.

Given the numerous challenges to establishing a cooperative agreement due to different valuations of hatchery-origin and wild fish, in the initial stages of development, such an organization could create principles to guide actions, which develop into more substantive regulations over time (Levy et al. 1992). For example, the principles could first discourage further increases in hatchery output, and then in subsequent amendments guide states toward regulations that limit hatchery releases using agreed-upon quotas,
taxes, or permits, and become stricter over time. The transition from principles to rules will require careful negotiation (Levy et al. 1992), considering the economic efficiency of proposed regulations (identified from a utilitarian perspective), changes in sovereignty over hatchery output (from a realist perspective), political feasibility and perceptions of equity (from an institutionalist perspective). Furthermore, by using ecological and economic information from collaborative research on the valuation of salmon and their prey and the effects of various regulations on hatchery output as focal points for action (Levy et al. 1992), collective action may become desirable for all members of the organization.

The third way that a new international organization could contribute to collective actions is to include sub-national governmental and non-governmental agencies on scientific assessment teams and in negotiations in order to increase domestic acceptance of any restrictions on hatchery releases. By incorporating views and preferences of these sub-national groups, negotiations may converge on a set of governing rules that national governments will be more willing to accept and better able to implement at the national level (a key concern of the institutionalist framework). Sub-national groups could also become a force operating across nations for change in international policies. If collective action is to be successful, incorporating these alternative views will be especially important for Japan, where concern about competition among salmon due to hatchery releases is much greater within the scientific community (e.g., Kaeriyama 1999), then among policy makers (X. Augerot, pers. comm. State of the Salmon, 721 NW Ninth Ave, Suite 280, Portland, Oregon, 97209). Furthermore, this institution could link international policy at the ocean-basin scale to hatchery-level regulations at regional- and local-scales.
by including stakeholders from industry and fisheries cooperatives (which are common in Japan; Vicuña 1991) in negotiations for a hierarchical arrangement of regulations (de la Mare 2005).

Fourth, a new organization could provide a system to monitor compliance with these regulations in order to prevent "free-riding". Such monitoring could be achieved through, for example, disclosing current hatchery data at regular meetings and in published reports, sharing of hatchery-related domestic policies, and random, on-the-ground "audits" of hatcheries. Furthermore, violations could be reported at international meetings and punished according to domestic legislation (as in the case of the NPAFC, where rights for prosecution are granted only to the country responsible for the violation, deReynier 1998).

Conclusions

By using three different theoretical frameworks (rational self-interest, realism, and institutionalism), I have developed a comprehensive overview of the incentives and disincentives for collective action on the problem of competition for prey resources by salmon and density-dependent reduction in growth in the North Pacific Ocean. From this multi-perspective analysis, I suggest the creation of an international organization with a mandate to maximize net benefits of hatchery releases combined across nation states, while ensuring an equitable distribution of net benefits among nation states. This mandate would explicitly recognize ecological interactions among species that share a common pool of prey resources. Such an organization, and the supporting agreement among relevant countries, could modify the incentive structure for collective action to reduce hatchery output and competition for prey resources. From a utilitarian perspective, side-
payments could be used to compensate states that would lose from such an agreement (e.g., Japan), either through financial transfers, funds directed to conservation of depleted wild stocks in Japan, or research and technology into methods to reduce competition among populations from different countries. From a realist perspective, changes in the character of sovereignty (including control, autonomy and legitimacy) could be negotiated to ensure that, for example, any losses in autonomy over hatchery output for one country coincide with gains from increased control over hatcheries releases from other countries. For example, for the United States and Canada, a loss of autonomy over domestic enhancement facilities for one country may be compensated by an increase that country's ability to restrict hatchery releases in other countries and therefore prevent further declines in body size of their own (United States and Canada) wild salmon populations. From an institutional perspective, negotiations that include notions of equity and establish a common framework for understanding the problem through cooperative research will facilitate collective action. Furthermore, when assigning access to the exploitation of shared prey resources by salmon (e.g., by restricting hatchery releases), lessons from other common-pool resources such as oil and ground water may help focus negotiations on essential criteria for cooperation (e.g., utilitarian costs and benefits and perceptions of equity). By creating such an organization, nation states of the North Pacific may be better equipped to avoid declines in salmon body size and associated losses in economic and ecological value of the fishery due to increased hatchery production.
Acknowledgments

Shane Frederick provided the original idea for this manuscript in a term paper at the School of Resource and Environmental Management, Simon Fraser University (1991). I would like to thank Xan Augerot (State of the Salmon, Portland, Oregon) for her insights about the North Pacific Anadromous Fish Commission.

References


Frederick, S. 1990. There's no such thing as a free lunch. School for Resource and Environmental Management, Simon Fraser University, Report for course REM 641, Policy Analysis.


Table 4.1 Incentives and disincentives for cooperation, as well as potential strategies for altering the incentive structure ("Strategies"), from a utilitarian perspective (columns 1 and 2). These are applied to the 1911 North Pacific Fur Seal Treaty and the problem of density-dependent growth of salmon in the North Pacific Ocean in the third and fourth columns, respectively.

<table>
<thead>
<tr>
<th>Row #</th>
<th>Utilitarian perspective</th>
<th>1911 North Pacific Fur Seal Treaty</th>
<th>Density-dependent growth of Pacific salmon</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>Incentives</td>
<td>1. Net benefits (increases in utility) from collective action that exceed those from unilateral action.</td>
<td>• For Russia and the US, the benefits derived from conservation of fur seal populations achieved from a ban on pelagic sealing were perceived to exceed the costs of foregoing the pelagic harvest; this was not the case for Japan and Canada, since they had no claims to terrestrial breeding grounds and would therefore gain no benefits from a ban on pelagic sealing (Barrett 2003).</td>
</tr>
<tr>
<td></td>
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<td></td>
<td>• Benefits from collective action to reduce density-dependent growth through reductions in total hatchery output may exceed costs of foregone hatchery production, at least for some countries.</td>
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<tr>
<td></td>
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<td></td>
<td>• Benefits are derived primarily from increases in commercial and recreational value associated with an increase in body size.</td>
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<td></td>
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<td>• In addition, increases in body size may result in increases in other use values (e.g., food and ceremonial value) and non-use values (e.g., existence and ecosystem values, Gende et al. 2002; Gislason 2006; Olson et al. 1991) if they are associated with higher reproductive output (eggs per spawner) and therefore larger population sizes of wild fish.</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>• Costs of reduced hatchery output are derived primarily from a decline in catch of hatchery-origin populations.</td>
</tr>
<tr>
<td>2</td>
<td>Disincentives</td>
<td>1. Uncertainty in identity of players, their interests, alternatives, and potential outcomes</td>
<td>• Identities and interests of players were known, and the biology of the resource was relatively well understood due to government research.</td>
</tr>
<tr>
<td></td>
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<td></td>
<td>• Alternative management actions were clear, the most prominent being a ban on pelagic sealing (Barrett 2003).</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>• Identities and interests of players are known.</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>• However, the relative extent of competition between salmon originating from different nation states and of different species for the common pool of prey resources is not clearly understood (Cooney and Brodeur 1998).</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>• Therefore, the range of potential outcomes from a reduction in hatchery output is uncertain.</td>
</tr>
<tr>
<td>Row #</td>
<td>Utilitarian perspective</td>
<td>1911 North Pacific Fur Seal Treaty</td>
<td>Density-dependent growth of Pacific salmon</td>
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<tr>
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</tr>
<tr>
<td>3</td>
<td>Disincentives cont.</td>
<td>2. Unequal value of costs and benefits due to differences in utilities or discount rates</td>
<td>- Costs of continued pelagic sealing and the benefits of banning pelagic sealing were larger for Russia and the US than for Canada and Japan (Dorsey 1991).</td>
</tr>
</tbody>
</table>
| 4     | 3. Lack of monitoring of compliance and enforcement ("free-rider" problem) | - No systematic monitoring of pelagic sealing was present prior to the 1911 treaty.  
- The treaty obliged states to monitor and enforce a ban on pelagic sealing after 1911 (Barrett 2003) (see "Strategies" section below). | - Currently, hatchery production in the North Pacific is reported by the North Pacific Anadromous Fish Commission, NPAFC. No institutional structure exists for regulating hatchery production or enforcing any regulations on hatchery releases. |
<p>| 5     | Strategies              | 1. Issue linkages                  | - There is a potential for countries to link density-dependent growth of salmon to other resource issues in the North Pacific, but these have not been explored. |
| 6     | 2. Side payments        | - Russia and the US agreed to give annual payments (in the form of 1000 seal skins or 15% of their annual harvest in cash, whichever was greater) to Canada and Japan, to compensate for the latter two countries' losses from foregoing pelagic sealing (Gay 1987). | - There is a potential to use side-payments to compensate states that would otherwise lose from an agreement to set quotas on total production (e.g., Japan) by those states that would gain (e.g., Canada). |</p>
<table>
<thead>
<tr>
<th>Row #</th>
<th>Utilitarian perspective</th>
<th>1911 North Pacific Fur Seal Treaty</th>
<th>Density-dependent growth of Pacific salmon</th>
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</thead>
</table>
| 7     | 3. Institutions to monitor compliance, create enforcement mechanisms, and prevent "free-riders" | - States with breeding populations were obliged to monitor pelagic sealing.  
- Each country was required to enforce legislation to punish cheaters domestically.  
- Entry of other countries was deterred by banning imports of seal skins by states not party to the treaty (Barrett 2003). | - A new international institution (similar to the NPAFC) could be created to regulate hatchery output, monitor compliance, and enforce regulations.  
- Threats of "free-riding" by other states are low since no other states have significant abundances of salmon that migrate to the North Pacific Ocean (Augerot 2005). |
<table>
<thead>
<tr>
<th>Row #</th>
<th>Realist perspective</th>
<th>1911 North Pacific Fur Seal Treaty</th>
<th>Density-dependent growth of Pacific salmon</th>
</tr>
</thead>
</table>
| 1     | Incentives          | 1. Pressure from a hegemon without explicit use of power | • No hegemon present.  
• In fact, in 1911, all four states were considered economic and political powers (Barrett 2003). | • No hegemon present (Kaye 2001). |
| 2     | Disincentives       | 1. Loss of power and prestige relative to other states | • In negotiations for a bilateral agreement between Russia and Japan in 1905, Russia perceived Japan as having little economic and political power in negotiations and offered Japan no compensation for a ban on pelagic sealing. Japan therefore resisted agreement initially because of fear of further loss of power and prestige relative to Russia (Dorsey 1991).  
• During negotiations for the 1911 treaty, Great Britain and Japan feared a loss of power and prestige relative to Russia and the US by foregoing pelagic sealing. | • Relatively symmetric balance of power among states (deReynier 1998). |
| 3     |                     | 2. Loss of sovereignty | • Great Britain and Japan also feared loss of sovereignty from a ban on pelagic sealing. | • All states may perceive a loss of sovereignty in their ability to artificially enhance their own stocks, but this perceived loss is likely largest for Japan (deReynier 1998). |
| 4     | Strategies          | 1. Structural power used for rewards or punishments | • The US and Russia agreed to reward Great Britain and Japan financially, or with seal skins, in exchange for a ban on pelagic sealing (Gay 1987). | • N/a (no perceived imbalance of power). |
| 5     |                     | 2. "Sovereignty bargaining" (i.e., trade-offs in components of sovereignty: autonomy, control, and legitimacy) | • For Great Britain and Japan, a loss of autonomy over the ability to harvest seals on the high seas was balanced by an increase in international legitimacy by partaking in the agreement and an increase in control over the resource (by using cooperative action to prevent further declines in abundance) (Young 1981). | • A loss of autonomy over domestic enhancement facilities for any one state may coincide with, for example, an increase in autonomy over enhancement practices in other states, and increased control over the change in body size of harvested salmon and reproductive output of stocks. |
Table 4.3 Same as Table 4.1, except from an institutionalist perspective.

<table>
<thead>
<tr>
<th>Row #</th>
<th>Institutionalist perspective</th>
<th>1911 North Pacific Fur Seal Treaty</th>
<th>Density-dependent growth of Pacific salmon</th>
</tr>
</thead>
</table>
| 1     | **Incentives**                | • Previous bilateral agreements prior to 1911 (Barrett 2003) had set the context for negotiations for the 1911 treaty.  
• The 1911 treaty deterred non-compliance by obliging states with breeding populations to monitor pelagic sealing, and enforce punishment of vessels that violate this ban (Barrett 2003).  
• The treaty also deterred non-participation by banning imports of "non-authenticated" seal skins (e.g., from states not party to the treaty) (Barrett 2003). | • Previous international agreements such as the Convention for the Conservation of Anadromous Stocks and the UN Agreement on Straddling Fish stocks and Highly Migratory Fish stocks have created norms of behaviour in international fisheries management (e.g., the precautionary approach, and accounting ecological interactions among species when setting regulations, Moore 1999; Tsuru 2004).  
• Furthermore, a new institution may prevent member states from "free-riding" by setting rules for hatchery output, monitoring compliance, and punishing cheaters (Joyner 1975, Frederick 1990; Heard 1998). |
<p>| 2     | <strong>Disincentives</strong>              | • The 1911 Fur Seal Treaty was desirable for all parties. After the stock collapsed in the early 1900s, cooperation was both feasible and desirable for all states because the value of harvest declined to such an extent that Japan and Canada no longer had incentives to continue pelagic sealing. Furthermore, with the compensation provided by Russia and the US, all four states had incentives to stop pelagic sealing entirely (Young 1994; Barrett 2003). | • Reductions in enhancement may be politically infeasible due to effects of facility closures on local economies and job loss. |</p>
<table>
<thead>
<tr>
<th>Row #</th>
<th>Institutionalist perspective</th>
<th>1911 North Pacific Fur Seal Treaty</th>
<th>Density-dependent growth of Pacific salmon</th>
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<tbody>
<tr>
<td>3</td>
<td>Disincentives cont.</td>
<td>Prior to the 1911 treaty, property rights over seal harvest were contested (Dorsey 1991).</td>
<td>Uncertainties exist over property rights of salmon prey in the North Pacific Ocean.</td>
</tr>
<tr>
<td></td>
<td>2. Uncertainty in preferences and outcomes of institutional arrangements (e.g., uncertainty in property rights)</td>
<td>Specifically, Russia and the US assumed exclusive property rights during all life stages for seals breeding in their respective territories, but Japan and Great Britain assumed that seals were common property to all states during their pelagic life stage (Dorsey 1991).</td>
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<td>After an intensive bargaining process, an agreement was reached that was perceived as fair by all states, in part by the use of compensation from the US and Russia given to Japan and Canada for foregoing pelagic sealing (Dorsey 1991), and by addressing contested property rights. Fur seals were designated a common property to all four states, with direct managerial authority given to the US and Russia (Young 1981).</td>
<td>Use institutional bargaining to focus attention on perceptions of equity and specific roles of states in domestic policy, instead of on absolute gains or losses in utility and relative gains or losses of power and sovereignty.</td>
</tr>
<tr>
<td>4</td>
<td>Strategies</td>
<td>The bargaining process was facilitated by the use of the focal point, &quot;a ban on pelagic sealing&quot; (Young 1994).</td>
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<tr>
<td></td>
<td>1. Use institutional bargaining to create &quot;contractual institutions&quot; that focus on non-utilitarian sources of behaviour (e.g., authority and role definition) or bargaining structure (e.g., unequal bargaining strength).</td>
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<td></td>
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<tr>
<td>Disincentives</td>
<td>Role of organization to remove disincentives</td>
<td></td>
<td></td>
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<td>--------------------------------------------------------------------------------</td>
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<tr>
<td>1. Scientific uncertainties about the causes of declines in body size and effects of reductions in hatchery output</td>
<td>• Create multi-national scientific assessment teams to resolve critical uncertainties and create a common frame of reference for negotiations</td>
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<tr>
<td>2. Unclear preferences and policy alternatives</td>
<td>• Negotiate policies to achieve the mandate given diversity in values among countries</td>
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<td></td>
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<tr>
<td></td>
<td>• Administer rules</td>
<td></td>
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<tr>
<td></td>
<td>• At first, voluntary principles could be endorsed, which develop into formal rules over time</td>
<td></td>
<td></td>
</tr>
<tr>
<td>3. Domestic infeasibility</td>
<td>• Include sub-national governmental and non-governmental agencies on scientific assessment teams</td>
<td></td>
<td></td>
</tr>
<tr>
<td>4. Free-riding (i.e., non-compliance of member countries to restrictions in hatchery releases)</td>
<td>• Monitor compliance through published reports, sharing of hatchery-related domestic policies, and random, on-the-ground surveys</td>
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<tr>
<td></td>
<td>• Punish cheaters through domestic legislation</td>
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</table>
Figure 4.1  Known range of immature Bristol Bay sockeye salmon and maturing eastern Kamchatka Peninsula pink salmon (from Ruggerone et al. 2003).
Appendix 4.1

I used two relations to calculate the value of an increase in body length equal in magnitude to the observed average reduction in length (1.3 cm) of Fraser River sockeye salmon from 1948-1994 (Pyper et al. 1999). The first was the relation between body length and body mass (D. Patterson, pers. comm., Fisheries and Oceans Canada, Cooperative Resource Management Institute at Simon Fraser University, Burnaby, British Columbia):

\[
m = \left(114.04e^{0.0622L}\right)/1000,
\]

where \(m\) is mass per fish (kg), \(L\) is the post-orbital hyperal length (cm), and \(e\) is the natural logarithm. The second relation was the price per kg of salmon paid to harvesters at the dock in 2003 ($4.00/kg) (Grant Snell, pers. comm, BC Salmon Marketing Council, 1100 – 1200 West 73rd Ave., Vancouver, British Columbia). Because price varies with gear type (e.g., value is higher for fish caught by troll than by seine), the total price was weighted by the proportion of fish caught with each gear type. I calculated the change in commercial value from a reference point based on current status of this population. Specifically, I used the total number and biomass of fish caught in 2003 (906 944 fish and 2 504 561 kg) (Grant Snell, pers. comm.), and a body length equal to 52.7 cm (the average body length observed for Fraser River sockeye salmon spawners from 1990 – 1994).
Summary and General Conclusions

This thesis quantified the magnitude and nature of a variety of sources of uncertainty that have previously been incompletely considered in stock assessment and fisheries management, and developed new methods to account for them. I found evidence for long-term trends in parameters of one forecasting model for sockeye salmon in British Columbia and Alaska, the sibling model (Chapter 1). However, accounting for these trends did not improve management performance when evaluated in a simulation model of the entire fisheries system (Chapter 3). Any changes in forecasting accuracy from accounting for those long-term trends in parameters of the forecasting model were either swamped by large interannual variability in ages-at-maturity or by uncertainties in other components of the fishery system. I also found evidence for imprecision and bias in the outcomes of implementing fishing regulations for Fraser River sockeye salmon (Chapter 2). When this source of uncertainty was accounted for in two sockeye salmon stocks (Togiak River, Alaska, and Chilko Lake, British Columbia), I found that management performance improved (Chapter 3). The results of this research will give managers a more realistic perspective on their ability to achieve harvesting and conservation objectives given biological and management uncertainties. Furthermore, it will help direct research efforts in reducing scientific uncertainties towards areas that will better assist managers in achieving their objectives. Finally, uncertainty in the ability to achieve collective action is a key challenge to addressing the problem of competition among salmon from different countries in the North Pacific Ocean that share a common
pool of prey resources. Although competition among salmon populations has been
described by previous authors, none has developed recommendations for cooperation. I
investigated the potential for international cooperation on this problem by identifying
incentives and disincentives for altering individualistic behaviour to encourage collective
action (Chapter 4). After considering three different theoretical perspectives on
international relations, I recommend creating a new international organization that,
among other functions, administers regulations in total output of hatchery fish across
salmon producing nations in the North Pacific, recognizing ecological interactions among
salmon populations and species that share a common pool of prey resources. This
organization could uses various tools such as international monitoring, multinational
scientific assessment teams, and side-payments to achieve cooperation.

This research reveals several remaining challenges for managing risks and
uncertainties in salmon fisheries, including the following, as explained in more detail
below:

(1) quantifying the magnitude and nature of outcome uncertainties for other fisheries not
analyzed here;

(2) accounting for the effects of interactions among uncertainties in management at
various spatial and temporal scales;

(3) characterizing structural uncertainties; and

(4) identifying objectives that represent different preferences for various attributes (e.g.,
commercial harvest and conservation).

Although, this thesis makes progress towards assessing the magnitude and nature
of outcome uncertainties for four stock aggregates in the Fraser River, British Columbia,
those patterns cannot be generalized to stocks in other regions. Unique biological and management characteristics may necessitate estimating stock- or region-specific parameters that describe outcome uncertainty (e.g., the transfer function used to simulate bias) in models for evaluating management actions. However, within regions, hierarchical Bayesian models can be used to identify prior distributions for parameters that describe this source of uncertainty for stocks with poor-quality (or no) data by "borrowing" information from neighbouring stocks. This approach has the advantage of incorporating parameter uncertainty into models that describe the deviations between management targets and outcomes.

Hierarchical modelling can also be incorporated into simulation models to account for interactions among uncertainties at different spatial and temporal scales of management control (challenge #2). Simulation models such as the one described in Chapter 3 that evaluate management alternatives can account for the cumulative effects of many sources of uncertainties from different components of the system. When these simulation models include multiple hierarchical levels of control, uncertainties at each level can also be accounted for. For example, management control over competition among salmon from different populations in the North Pacific Ocean can be divided into several hierarchical levels: international polices to address these interactions that occur at an ocean-basin scale; actions by national management agencies to comply with international policies that occur at the national scale; and local control over hatchery production by the hatchery facilities themselves. Hierarchical levels are linked by objectives and performance metrics (i.e., the performance metrics at higher levels shape the objectives at lower levels), and uncertainties may occur when implementing
regulations at each hierarchical level. In this framework, links among hierarchical levels can be made explicit, which facilitates the propagation of uncertainties among scales in simulation models that evaluate management options.

Adaptive management is a tool commonly used to resolve parameter uncertainty, but it can also be used for addressing structural uncertainties (challenge #3) either passively through collection of data over time and updating of models, or actively, through experimentation (Walters 1986). This approach has the advantage of linking processes that occur at short temporal scales (from regular collection of new data) to those that occur over longer periods at the scale of management experiments. Another approach to addressing structural uncertainties is using sensitivity analyses on assumptions about the structure of the system in simulation models to identify management options that are robust to those assumptions. Furthermore, the precautionary approach to management can be used to prevent irreversible, harmful outcomes from management actions.

The fourth challenge, identifying management objectives that represent different preferences among stakeholders, can be developed through an iterative process of negotiations among stakeholders and implementation of regulations. Negotiations should involve utilitarian costs and benefits and perceptions of equity among diverse users. Simulation models, such as those presented in this thesis, can be used as a tool to negotiate trade-offs in preferences by simulating potential consequences of management actions and presenting performance metrics that represent different preferences. These models can help focus discussions among stakeholders on mutually acceptable, realistic outcomes.
To conclude, this thesis provides the tools necessary to evaluate management alternatives in a more realistic way than has been done in the past. Accounting for key uncertainties in biological and management components of the fisheries system using the methods presented here can guide decision makers towards management actions that will better achieve objectives.

References