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Estimating the Impacts of Chinook Salmon Abundance and Prey Removal by Ocean Fishing on Southern Resident Killer Whale Population Dynamics

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Northwest Fisheries Science Center

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ii

Table of Contents

iv

List of Figures

List of Tables

viii

Executive Summary

This technical memorandum describes modeling analyses done for a series of three bilateral workshops investigating the effects of Chinook salmon (*Oncorhynchus tshawytscha*) fishing on Southern Resident killer whales (SRKW, *Orcinus orca*). The workshops were held September 2011 in Seattle, Washington, March 2012 in Vancouver, British Columbia, and September 2012 in Seattle. They were jointly convened by the National Marine Fisheries Service (NMFS) and the Fisheries and Oceans Canada in order to evaluate scientific information and analyses being used by NMFS to evaluate the effects on SRKWs of prey reduction by salmon fisheries. Both the whales and their salmon prey inhabit transboundary waters, thus effective management of both species is a bilateral issue. Material presented during these workshops was critically evaluated by a panel of prominent biologists with expertise in ecology, fisheries science, and marine mammal biology, with the goal of evaluating the strength of evidence and inferences presented by the agencies and their partners.

The SRKW population is listed as endangered under the Canada Species at Risk Act (2003) and U.S. Endangered Species Act (2005). The population is relatively small, ranging from 71 individuals in 1974 to a maximum of 97 individuals in 1996 (based on the July census). The workshops focused on the population status through 2010, but as of July 2012, the SRKW population numbered 85 animals. Previous studies have shown that indices of Chinook salmon are correlated with SRKW population dynamics, and that prey samples collected at feeding events are predominately composed of Chinook salmon.

SRKWs have been shown to have lower growth rates than a neighboring population of resident whales, and using slightly different statistical methods, we confirmed those results with a longer data set. The differences in growth rates between the Southern Resident and Northern Resident populations are due to differences in fecundity and survival between populations. The exact mechanisms responsible for creating density dependence in killer whales are unknown; however, we found that the strength of density dependence appears to be stronger in the Northern Resident population. Southern Residents may be at or near carrying capacity, but because the population has experienced relatively small fluctuations, estimating carrying capacity is uncertain. Another reason why estimating density dependence for Southern Residents may be so difficult is that carrying capacity may be related to killer whale prey availability, which is variable over time.

As with previous analyses, we found strong correlations between killer whale demographic rates and indices of Chinook salmon abundance. In our consideration of the new stock-specific Chinook salmon indices, the high correlation between stock indices over time prevented a single stock or group of stocks from being strongly identified as most correlated with Southern Resident population dynamics. Of all groups considered, a seasonal fall group not including California stocks was most supported by the data; however, in subsequent analyses, a larger aggregate of all stocks performed slightly better. While these types of correlative analyses

are important for identifying possible relationships, there is no direct link between correlations and causal mechanisms. In other words, the Chinook salmon stock or group of stocks that is most correlated in these types of analyses is not necessarily the stock or group that is limiting SRKW growth or recovery. Complicating matters, the relative importance of individual stocks likely changes over time.

In a retrospective analysis of potential decreases in prey availability as a result of fishing, we found that if fishing had been reduced at any point in the time series, the largest potential benefits would have been early in the time series (early 1980s) or in years when Chinook salmon indices were low (late 1990s). In recent years, reducing Chinook fishing likely has a smaller impact, because exploitation rates in ocean fisheries have generally declined since the 1970s. For future projections under alternative amounts of Chinook availability, we found that the Southern Resident population will likely have a slightly positive growth trajectory, but will likely not meet recovery goals in the next 30 years. For example, even if average Chinook abundance were increased to the maximum that has been observed historically (1979–2010), the probability of the Southern Resident population being ESA-downlisted to threatened status is less than 60%. Because of uncertainties in the demographic relationships between killer whales and Chinook salmon, and uncertainty resulting from demographic stochasticity, our projections become highly uncertain across all levels of hypothetical salmon abundance, with coefficients of variation exceeding 200% over 5 years. Therefore, even though reducing Chinook salmon fishing might have some short term benefit to killer whales, there is low confidence in our ability to detect any difference resulting from this increase in prey.

The objective of this technical memorandum is to summarize some of the quantitative analyses and relationships investigated during the workshops, with particular emphasis on the analyses in which NMFS played a large role. This document is not meant to be exhaustive or discuss all uncertainties in different data sets used. For more details, readers are directed to The Effects of Salmon Fisheries on Southern Resident Killer Whales: Final Report of the Independent Science Panel, published by NMFS and Fisheries and Oceans Canada.

Introduction

The Southern Resident killer whale (SRKW, *Orcinus orca*) population is one of several killer whale populations inhabiting waters along the west coast of the United States and Canada (Olesiuk et al. 1990, Ford et al. 2000, Krahn et al. 2002, Krahn et al. 2004). Since the 1970s, researchers have identified several distinct ecological forms of Pacific killer whale, including fish-eating residents, mammal-eating transients (also known as Bigg's killer whales), and offshores (Ford et al. 2000). Photo identification methods have allowed researchers to identify and track the fates of individual animals within populations, allowing for detailed demographic analysis (Bigg et al. 1990, Olesiuk et al. 1990, Krahn et al. 2004, Olesiuk et al. 2005, Ward et al. 2009a). In addition to demographic monitoring, considerable work has been conducted on the behavior (Noren et al. 2009, Ashe et al. 2010, Noren and Hauser 2012), social structure (Parsons et al. 2009, Pilot et al. 2010, Beck et al. 2012), movements (Hauser 2006), and diet (Ford et al. 1998, Ford and Ellis 2006, Newsome et al. 2009, Hanson et al. 2010, Williams et al. 2011) of Pacific killer whales.

The analyses described here focus primarily on the SRKWs. This population is the most southerly distributed of several eastern North Pacific fish-eating killer whale populations (Krahn et al. 2002). Concern over the small SRKW population size and a sharp decline from 97 animals in 1996 to 79 animals in 2001 led to SRKWs being listed as endangered under the Canada Species at Risk Act in 2003 and the U.S. Endangered Species Act in 2005 (Krahn et al. 2004). For purposes of comparison, the analyses described here also include a second population, the Northern Resident killer whales (NRKW). After the SRKW, the NRKW population is the next most southerly distributed fish-eating killer whale population in the eastern North Pacific, and is ecologically very similar to the SRKW, particularly with regard to sharing much of the same diet and prey base. However, the NRKW population is considerably larger, with a population size of greater than 250 in 2012, compared to 85 for the SRKW, and has experienced a higher rate of population growth since the 1970s. The two populations have overlapping ranges (Ford et al. 2000), with the SRKW range extending from California to Southeast Alaska and the NRKW range from central Washington to Southeast Alaska (Ford et al. 2000, Krahn et al. 2002). Over the last 30 years, the NRKW and SRKW populations have experienced correlated population dynamics, typically experiencing good and bad years at the same time (Ford and Ellis 2005, Ford et al. 2010), but have remained genetically distinct, with no known exchange of individuals (Krahn et al. 2004, NMFS 2011b).

Several risks have been identified as potentially threatening the SRKW with extinction. When the SRKW and NRKW populations were first subject to serious investigation starting in the 1970s, their numbers were known to be reduced from historical levels (Krahn et al. 2002). Prior to 1974, both the SRKW and the NRKW populations were subject to a live capture fishery to catch animals for display in aquaria (Bigg and Wolman 1975). In addition, due to lack of legal protection and a tendency to interact with fisheries, the populations were likely to have been subject to some unknown level of direct mortality prior to implementation of the Marine

Mammal Protection Act in 1972 (Krahn et al. 2002). In addition to direct mortality prior to the 1970s, a number of other risk factors have been identified as current threats to the SRKW (Krahn et al. 2004, NMFS 2008). Among these are prey limitation, inbreeding as a result of the small population size (Ford et al. 2011), chemical contaminants (Krahn et al. 2007), acoustic disturbances or other negative effects of whale watch or commercial vessels (Holt et al. 2009, Lusseau et al. 2009, Lachmuth et al. 2011), other anthropogenic disturbances, and catastrophic events such as oil spills (Krahn et al. 2002, Krahn et al. 2004). Evaluating the relative magnitudes of these diverse threats is difficult, due to both limited direct data on some threats and widely different modes of action. Where long-term time series data related to particular threats do exist, only abundance of prey has been found to be directly correlated with SRKW demography (Ward et al. 2009a).

In 2011 the National Marine Fisheries Service (NMFS) issued a biological opinion in response to a fishing plan proposed by the Washington Department of Fish and Wildlife and the Puget Sound Treaty Indian Tribes that would govern their Chinook salmon (*Oncorhynchus tshawytscha*) fisheries in Puget Sound. The analysis of the proposed Puget Sound harvest plan included a retrospective look at the effects on harvest, exploitation rates, and abundance of Chinook salmon returning to spawn—if the plan had been in effect during 1994 through 2008. This analysis was also used to estimate the effect on kilocalories of food energy of Chinook salmon prey for SRKWs in inland waters during July through September under three levels of fishing season structure: no salmon fishing on the U.S. and Canadian west coast, no salmon fishing in U.S. coastal waters, and no fishing within Puget Sound. The Puget Sound analysis also incorporated new information on correlations between certain indices of Chinook salmon abundance and the SRKW population growth, as well as a broader range of years than previously had been used to represent interannual variations in Chinook salmon abundance.

Because of the potential ramifications of these analyses to salmon fisheries management and other activities affecting the abundance of Chinook salmon, NMFS and Fisheries and Oceans Canada sought to ensure that the scientific information used to inform management decisions is the best available. Toward this end, the agencies jointly commissioned an independent, multidisciplinary panel of expert scientists to review the available science in an open and inclusive scientific workshop process. The key question evaluated in these workshops was:

> To what extent are salmon fisheries affecting recovery of SRKWs by reducing the abundance of their available prey, and what are the consequences to their survival and recovery?

A series of 3-day scientific workshops was attended by all seven members of the Independent Science Panel and nearly a hundred other invited scientists representing NMFS, Fisheries and Oceans Canada, academia, industry, conservation organizations, management agencies, tribes, First Nations, and others. The first workshop occurred September 21–23, 2011, in Seattle, Washington, and the second March 13–15, 2012, in Vancouver, Canada. The panel issued a draft report following the second workshop. Comments received on the panel's draft report and other analyses requested by the panel were presented and considered at the third workshop, held September 18–20 in Seattle. The panel issued its final report on November 30, 2012. Interested parties were encouraged to provide any comments in response to the report by January 31, 2013. Additional details describing the workshop process and the agendas, the pertinent scientific

literature, and scientific presentations considered at the workshops can be found online.[1] In interpreting the results presented in this document or the Independent Science Panel report, it is important to highlight that the panel was not asked to perform analyses or make management decisions; instead, it was only asked to critique and evaluate all of the available science.

Two types of information suggest that SRKWs 1) depend on Chinook salmon, and 2) may be prey limited. First, prey and fecal samples have been collected during observed predation events and later identified to the species and population level (Ford and Ellis 2006, Hanson et al. 2010). Because of logistical difficulties associated with prey sampling, these samples are concentrated in summer months and are largely from the relatively protected waters of the Salish Sea. Diets inferred from these studies are similar between NRKWs and SRKWs, with Chinook salmon representing the majority of recovered prey: 72% of scales and tissue in Ford and Ellis (2006) and 76% of scales and tissue in Hanson et al. (2010). Stable isotope data from resident killer whale blubber is also consistent with a diet dominated by Chinook and other salmonids (Herman et al. 2005). The second type of analysis supporting the link between Chinook salmon abundance and killer whale demographics relies on correlative studies between estimated annual population abundance of salmon and observed killer whale births and deaths (or birth rates and death rates) (Ford et al. 2005, Ward et al. 2009a, Ford et al. 2010). Though each of these studies has relied on different salmon indices and uses different responses (e.g., individual births versus population birth rates), each has found significant positive correlations between years of high Chinook salmon abundance and years of high SRKW population growth. Additional evidence that may support the prey limitation hypothesis are the anecdotal observations of SRKW individuals appearing to be in a starved condition ("peanut-head" syndrome) (Durban et al. 2009, Hilborn et al. 2012), and recently developed hormone indicators that may indicate seasonal periods of nutritional stress (Ayres et al. 2012). In this report, we focus on the details of the correlative studies, which made up the majority of the work for the bilateral workshops (Hilborn et al. 2012).

In addition to knowing SRKW births and deaths at the population level, individual resighting histories are known, because every individual in the population is recognizable based on unique natural marks that remain with it over the course of its life (Bigg et al. 1990, Ford et al. 2000, Center for Whale Research 2010). Since 1979 the SRKW population has been surveyed annually and, because every animal has been seen each year (or presumed dead if missing for more than a year), this annual survey represents a complete population census (Center for Whale Research 2010). The NRKW population has been surveyed annually over the same time period (Ellis et al. 2011). In addition to births and deaths, the maternal line of every individual is also known, due to the cohesive social structure of resident killer whales (Bigg et al. 1990, Parsons et al. 2009, Foster et al. 2012).

In ecology and especially in the aquatic sciences, data sets recording more than 30 years of individual births and deaths are rare. Though unique in its length and detail, the SRKW data set also presents some interesting statistical challenges. First, SRKWs are extremely long-lived (some females are thought to have reached 80–90 years of age) (Center for Whale Research 2010). It is much more challenging to estimate interactions between environmental drivers and

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[1] Information on the workshops, including copies of all presentations, is online at http://www.nwr.noaa.gov /protected_species/marine_mammals/killer_whale/effects_fisheries.html.

population growth when species are long-lived relative to species that are much shorter lived (Ozgul et al. 2009). Second, because the SRKW population is small and events used to estimate demographic rates (births, deaths) occur relatively infrequently, estimated correlations between environmental variables and demographic rates are inherently more uncertain than they would be for a larger population (Coulson et al. 2001, Stenseth et al. 2004). Statistically, this also means that models for binary data at the individual level (Cooch and White 2006) have to be used over standard regression tools for continuous responses.

Finally, because the SRKW population inhabits such a large range (central California to Southeast Alaska), conducting experimental studies to verify correlations are nearly impossible—meaning that the exact mechanisms linking prey availability to resident killer whale demography will likely remain unknown. This problem is not unusual in ecology or in marine fisheries research. One of the best examples of experimental manipulations in marine ecology involves research in the rocky intertidal; these experiments work well on very small spatial or temporal scales, but are also impossible across large scales (Underwood 2000). Conducting experiments on SRKWs and their prey would ultimately involve manipulations of multiple ecosystems across the North Pacific, a difficult task for ecologists even at the scale of individual ecosystems (Schindler 1998).

The purpose of this report is to document the details of the population modeling (both Chinook salmon and SRKWs) that were presented to the Independent Science Panel during the workshop process. The workshops included presentations on a wide variety of topics, such as diet, whale condition measured through photographs and hormones, contaminant profiles, and many others.[2] Our focus, however, is solely on documenting the demographic models used to 1) understand correlations between Chinook salmon abundance and SRKW and NRKW population dynamics, and 2) use those statistical relationships to explore the hypothetical effects of changing salmon abundance on the SRKW population trends. While this report is meant to be a stand-alone document, many of the questions and hypotheses evaluated here reflect analyses that were done to address specific questions asked by the panel. Further details about the motivating questions are presented in the Independent Science Panel report (Hilborn et al. 2012).

The remaining sections of this technical memorandum are as follows: in the Summarizing Population Dynamics of SRKWs section, we present some overall patterns and trends in SRKW population dynamics over the last 30 years; in the Estimating Differences in Survival, Fecundity, and Growth Rates section, we provide details on several approaches that were used to estimate the carrying capacity of SRKWs; in the Calculating Stock-specific and Aggregate Salmon Indices section, we have an overview of how SRKW population growth has performed relative to NRKWs; in the Estimating Carrying Capacity and Strength of Density Dependence section, we provide details for how stock-specific Chinook salmon time series are generated; in the Identifying Salmon Stocks section, we indicate which of those Chinook stocks are most correlated with SRKW population growth; in the Estimating Historical Effects section, we estimate the historical effects of ocean Chinook fishing on SRKW population growth; and in the Estimating Future Effects section, we estimate future effects of changes in Chinook salmon abundance on SRKW population growth. Two appendices are also provided, one on SRKW population summaries, the other on output from SRKW population projections.

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[2] See footnote 1.

Summarizing Population Dynamics of SRKWs

Before conducting any population modeling or statistical comparisons between SRKWs and NRKWs, we performed basic population summaries, focusing on SRKWs. Previous work has documented trends and patterns in NRKWs (Olesiuk et al. 2005), as well as differences between these two populations (Olesiuk et al. 1990). Inference from these analyses has provided support for several important life history characteristics: 1) resident killer whales are long-lived, 2) as with many large terrestrial and marine mammals, female killer whales have a higher life expectancy than males, 3) like other long-lived mammals, female killer whales experience reproductive senescence, and 4) survival curves for both male and female killer whales are U-shaped, with the majority of mortalities occurring for young and old individuals (Olesiuk et al. 2005).

The SRKW population, numbering 85 animals in the July 2012 census, is divided into three subgroups or pods (referred to as J Pod, K Pod, L Pod). Each pod is comprised of multiple matrilines (a female and any offspring she has, as well as any offspring they may have). The social structure of killer whales is relatively static in that, from birth, individuals remain associated with their matriline and usually also their pod for the entire course of their lives (Ford et al. 2000, Parsons et al. 2009), so it was important to summarize population quantities for the entire SRKW population as well as at the pod level. We summarized births and deaths, as well as birth rates and death rates, by pods, sex, and age-class (Appendix A). The main point of this exercise was to create visual summaries that could later be used to generate hypotheses for other analyses (such as looking for evidence of density dependence). Since 1979 the SRKW population has experienced slightly more births than deaths (Table 1, Appendix A) and thus has a slightly positive net growth rate. No obvious temporal trends in birth or death rates exist, either by pod or for the aggregate SRKW population.

Part of the reason why the SRKW population size is small is that, from 1962 to 1973, it was targeted for live capture removals. Though NRKWs were also targeted, the majority of captures were taken from the SRKW population. Approximately 47 animals were removed before 1974 from the SRKW population, nearly all of them juveniles (<10 years old) (Bigg and Wolman 1975, Hoyt 1990, Ford et al. 2000). To examine trends related to legacy effects from these removals, we calculated the number of young (<21) and young or reproductively mature (<43) animals in each of the three SRKW pods (Figure 1 and Figure 2). These plots illustrate that one of the most serious trends affecting SRKWs is the change from a population initially biased toward females at the start of the time series to one that is biased toward males. Of the three pods, J Pod is biased toward females, but K Pod and L Pod are biased toward males. For example, of those individuals less than 43 years old, nearly 50% are female in K Pod and L Pod (55% and 50%, respectively), but 71% are female in J Pod (Figure 2). For the younger population segment (<21), K Pod and L Pod are 36% female, and J Pod is 63% female (Figure 1). These trends and the current age distribution of females in the SRKW population (Table 1) suggest that future population growth for the SRKW population will be driven by females in J Pod. It is important to also note that these legacy effects of removals would be expected to

possibly impact the total births at the population level (by reducing the number of reproductive females), rather than affect the birth rate of individual females. There is no suggestion of a postharvest decrease in total SRKW births (Appendix A).

Table 1. SRKW female ages by pod for females less than or equal to 42 years old (at the end of 2010). Data collected by the Center for Whale Research and NMFS.

J Pod	K Pod	L Pod
1	1	1
1	3	8
4	6	16
6	17	16
6	24	18
8	25	20
10	26	21
11	33	21
13	39	24
15	—	25
16	—	34
18	—	34
26	—	34
32	—	37
34	—	—
37	—	—
39	—	—

6
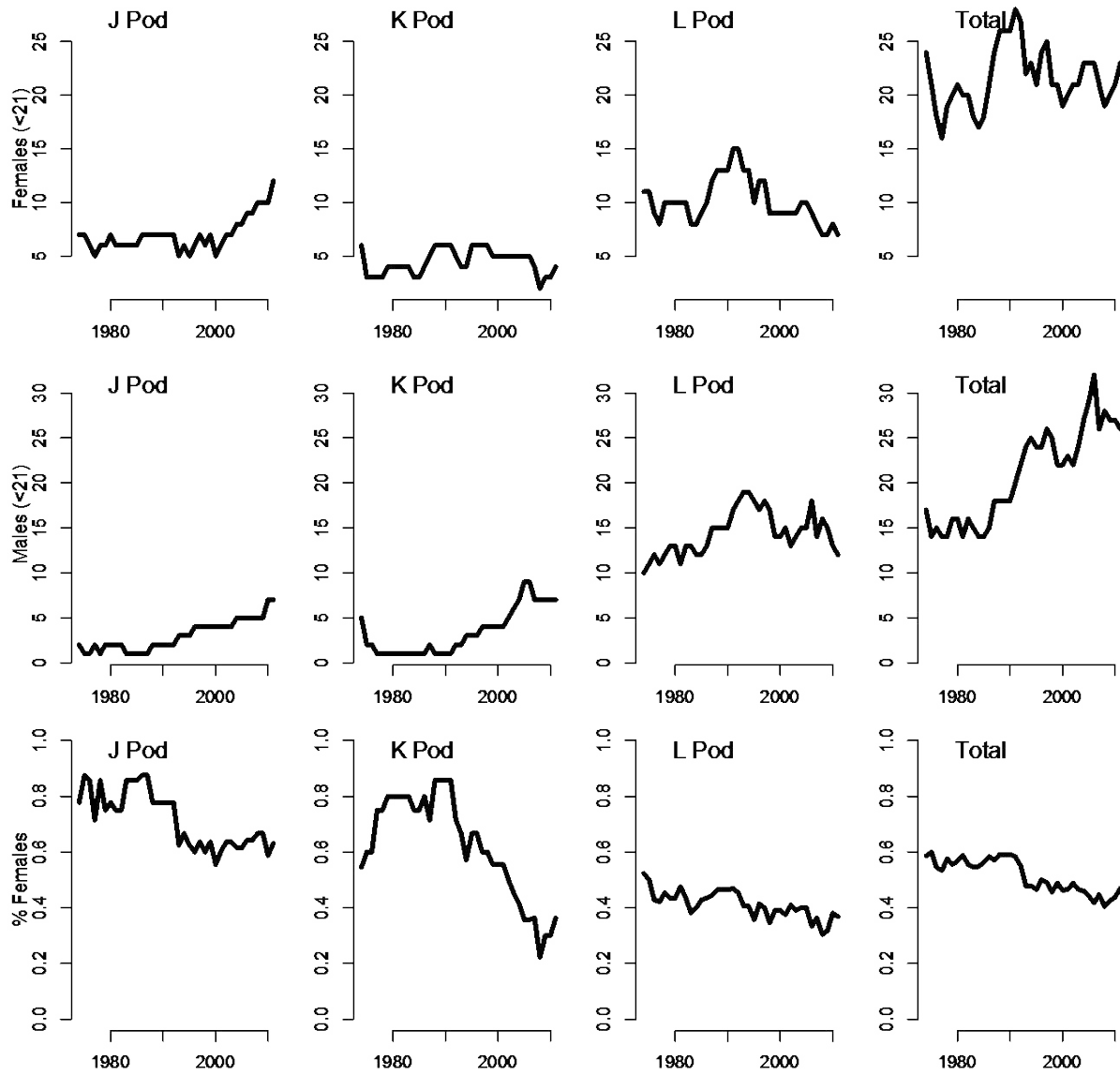
Figure 1. Summaries of young females, young males, and the sex ratio (percent female) of young animals in the SRKW population, 1974–2011.

7
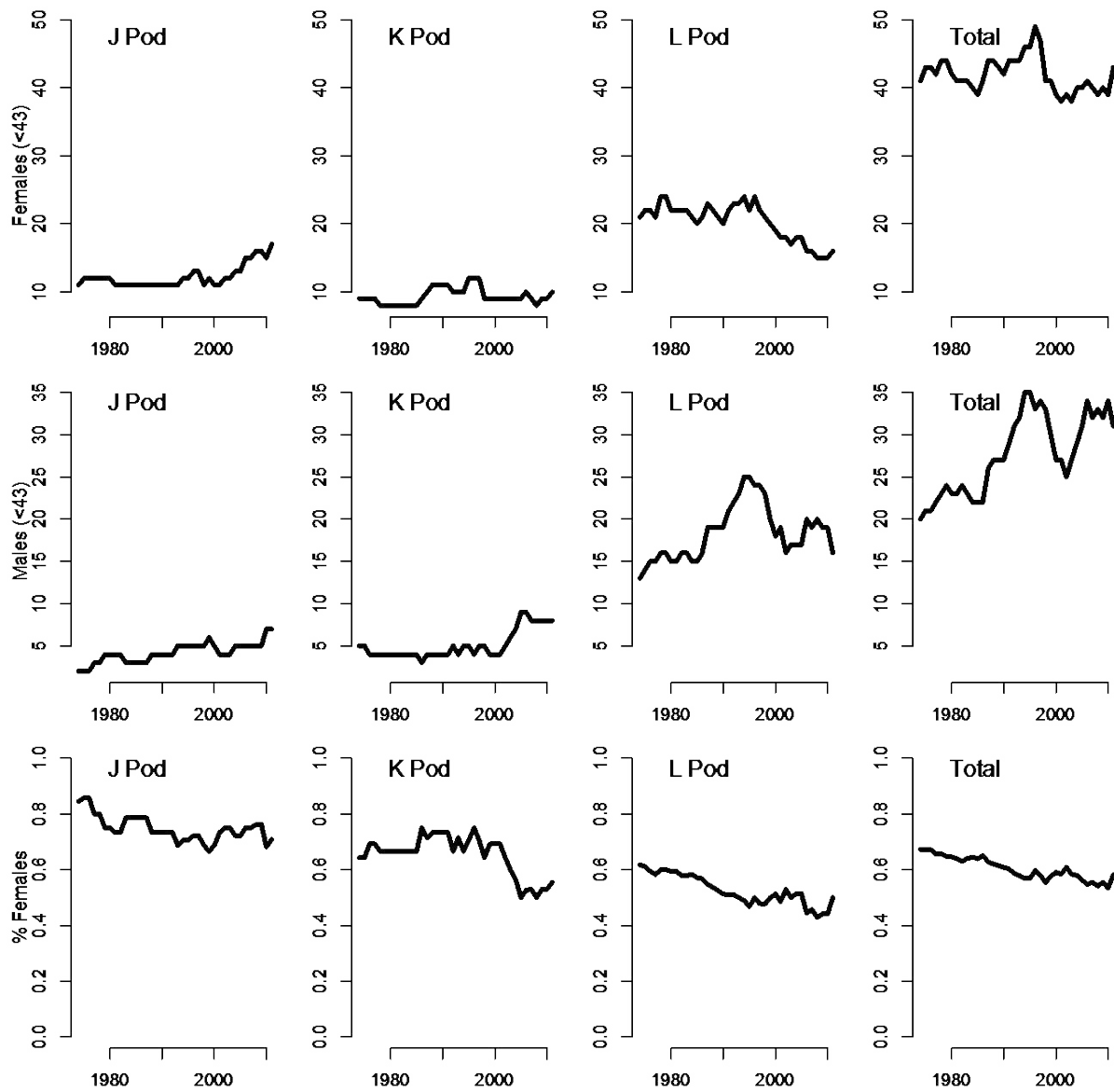
Figure 2. Summaries of young or reproductive females, young or reproductive males, and the sex ratio (percent female) of young or reproductive animals in the SRKW population, 1974–2011.

8

Estimating Differences in Survival, Fecundity, and Growth Rates between NRKWs and SRKWs

Because the SRKW data set represents a complete population census, it is equivalent to a mark-recapture study with a perfect detection probability (Cooch and White 2006). For the purposes of the workshops and this report, we can assume independence between models of fecundity (birth rates) and models for survival (death rates). Biologically, this means individuals who have recently given birth do not have reduced survival. Because of the low variability in the SRKW data, analyses of demographic rates and effects of covariates (density dependence, prey indices) can be improved by combining the SRKW census data alongside Fisheries and Oceans Canada's NRKW data. In most recent years, all individuals in the NRKW population have been seen; however, because this criteria has not been met in all years, the survey is not technically a complete census. For this combined analysis with the two populations, scientists from Fisheries and Oceans Canada standardized the SRKW data set to periods of the NRKW survey.

First, we treated SRKW and NRKW fecundity as the response variable. Following Ward et al. (2009a), five SRKW females with estimated age at the start of the study based on their last observed birth (K3, L2, L3, L11, and L21) were removed from the analysis to avoid introducing bias. The remaining births from 1980 to 2011 were included in our analysis and coded as a binary response (birth, no birth). Years immediately before and after observed births were treated as missing data, because with an 18-month gestation time, it is physically impossible for killer whale females to give birth in successive calendar years. The observed birth data was linked to population covariates via a link function (equivalent to logistic regression),

$$\text{logit}(f_{i,t}) = B_{0,pop} + B_1 \text{age}_{i,t} + B_2 \text{age}_{i,t}^2 + B_3 \text{age}_{i,t}^3 + B_4 \text{age}_{i,t}^4 \tag{1}$$

where $f_{i,t}$ represents the probability of animal i giving birth at time t, the coefficient $B_{0,pop}$ is an intercept allowing SRKW and NRKW to have different mean fecundity rates, and B_1, B_2, B_3, B_4 relate the effect of age on the probability of giving birth. The observed binary data is then modeled as $Y_{i,t} \sim Bernoulli(f_{i,t})$. It is important to note that covariates affect birth rates linearly in link space, but the relationship becomes nonlinear in normal space via the inverse link function. Our fecundity models were initially fit with age alone (no external covariates), then modified to include effects of Chinook salmon indices and density dependence.

Second, we examined support for density dependence in survival rates. Survival rates ($\phi_{i,t}$) were also modeled with a link function as

$$\text{logit}(\phi_{i,t}) = B_{0,pop} + B_{1,\text{stage}} + B_2 L_{i,t} \tag{2}$$

where $B_{0,\text{pop}}$ represents the population effect (NRKW, SRKW), $B_{1,stage}$ estimated stage-specific survival rates, B_2 is an extra intercept parameter allowing mean survival rates for L Pod to be slightly different, and $L_{i,t}$ is a binary indicator covariate specifying whether each individual is a member of L Pod. Previous work has estimated survival rates of L Pod to be slightly smaller

9

than those of J Pod or K Pod (Ward et al. 2011). When we applied this model to the SRKW data set alone, the population effect was not included in the model. There are several important differences in the models for survival and fecundity. First, because of uncertainty in the ages of some older individuals at the start of the 1979 census, survival was estimated as stage based rather than age based. Stages were slightly modified versions of those presented in Krahn et al. (2004): calves (age 0 animals seen in their first summer), juveniles (both sexes, animals aged 1–9), young reproductive females (ages 11–42), young males (ages 10–21), old postreproductive females (43+), and older males (22+). As with fecundity, we excluded several whales from the analysis to avoid potential biases: the deaths of L97, L98 (Luna), and L104 were not included. L98 was included in survival estimation of juveniles, but his death was not included because it was unnatural (ship strike). The deaths of L97 and L104 were also not included, because they were calves that disappeared at the same time as their mothers (thus their deaths are not independent). Though the deaths of these three individuals were not included in our estimation, their births were included in estimates of fecundity.

In addition to evaluating requests from the science panel for calculating estimates of demographic rates (Hilborn et al. 2012), we calculated estimates of the population growth rate (lambda, λ) that depends on these rates. Statistically, this involved turning estimates of fecundity and survival from individual resighting histories into a demographic transition matrix, and estimating λ as the dominant eigenvalue of that matrix (Fujuwara and Caswell 2002). Another way to interpret λ is in terms of R, the discrete net growth rate ($\lambda = 1 + R$), or in terms of r, the instantaneous rate of increase ($\lambda = \exp(r)$). These variables can also be linked back to the NMFS recovery plan, where the maximum growth rate observed over a 12-year period has been calculated as R = 0.023 (= ln(97/74)/12, Table 2). Because both our modeling approach and the data differ from the matrix-modeling approaches that have been used previously for these populations (Olesiuk et al. 1990, Brault and Caswell 1993), we expect our results to also differ slightly. First, we used Bayesian logistic regression to estimate the posterior distributions of the fecundity and survival models. We saved 25,000 vectors of Markov Chain Monte Carlo (MCMC) draws, and used the Gelman-Rubin diagnostic to indicate convergence (Gelman and Rubin 1992). While λ is a useful summary of population growth, it also has several limitations. First, it represents the average growth rate that is expected for a population at equilibrium; for small populations like SRKWs that are constantly affected by stochastic births and deaths, the population will never be at the exact equilibrium. Second, our growth rate estimate only includes females that are less than 43 years old; postreproductive females and all males are not included in the model and have no impact on λ (Caswell 2001).

Previous work has estimated significant differences in both demographic rates and population growth rate between SRKWs and NRKWs (Ford et al. 2005, Ward et al. 2009a), and those results were confirmed in this update. For fecundity modeling, the posterior distribution of the population effect indicated that there is a 99.7% chance that SRKW fecundity is lower than NRKW fecundity (Figure 3 and Figure 4). For survival modeling, the posterior distribution of the difference indicates that there is a 97.1% chance NRKW survival is higher than that of the SRKW (Figure 5). These differences in survival rates can also be interpreted in terms of the mean rates (Figure 6) or in terms of how these rates affect lifetime expectancy. Median female life expectancy in the SRKW population based on these rates is 37.8 years, versus 44.9 years in the NRKW population. Similarly, the median male age is 22.5 years in the SRKW population and 24.7 years in the NRKW population. Linking fecundity and survival, another way to express

10

the differences in fecundity rates is in terms of how many calves a female killer whale produces over her lifetime. Incorporating population and stage-specific survival rates (Figure 6), we performed a Monte Carlo simulation with the estimates of the fecundity model (Figure 4). This step was necessary, instead of just summing rates over all ages, because females are not able to produce calves in sequential years (Ford et al. 2005), and experienced natural mortality. Over a female's lifetime, this simulation approach indicates that a SRKW female would be expected to produce 3.1 calves, versus 3.5 in the NRKW population. For a female that lives to reproductive senescence, this difference is even greater (3.9 vs. 5.1).

Because fecundity and survival rates in the SRKW population are lower than in the NRKW population, it is not surprising that our estimates of population growth (λ) are also different between the two populations (Figure 7). The mean estimate of λ for SRKWs is 1.01 (95% credible intervals = 0.997,1.023) and the mean estimate of λ for NRKWs is 1.028 (95% credible intervals = 1.016,1.039). In interpreting λ, remember that λ represents the average replacement of females by females for a population at equilibrium. It does not incorporate stochastic variation from year to year or the observed total SRKW or NRKW population sizes, the latter being what the NMFS recovery criteria are based on (NMFS 2008).

Mechanisms responsible for the difference in growth rates between populations remain largely unknown. One hypothesis is that SRKWs may be at the southern edge of a latitudinal gradient for the resident fish-eating ecotype. Differences in growth rates between segments of NRKWs and SRKWs have been reported previously (Ford et al. 2000). To quantify variability among different levels of social structure, we replicated our logistic regression analyses, including random effect terms either at the clan, pod, or matriline level. Matriline effects are difficult to estimate for some segments of the populations; however, there is large variability at the pod level (Table 3). When the random effects are included at the clan level, the Southern Resident population has the lowest fecundity and survival rates; of the Northern Resident population clans, G Clan (which is distributed more southerly and has a range most overlapping Southern Residents) has lower survival and fecundity than clans with more northern distributions (A Clan, R Clan, Table 3).

Table 2. Annual census numbers from surveys. Data collected by the Center for Whale Research and NMFS. Numbers are typically reported July 1 and at the end of the calendar year; these numbers are the SRKW population size at the end of the calendar year (NMFS 2008).

Year	Population	Year	Population	Year	Population
1974	70	1987	84	2000	83
1975	71	1988	85	2001	81
1976	70	1989	85	2002	83
1977	79	1990	89	2003	84
1978	79	1991	92	2004	88
1979	81	1992	91	2005	88
1980	83	1993	97	2006	90
1981	81	1994	96	2007	87
1982	78	1995	98	2008	85
1983	76	1996	97	2009	87
1984	74	1997	92	2010	86
1985	77	1998	89		
1986	81	1999	85		
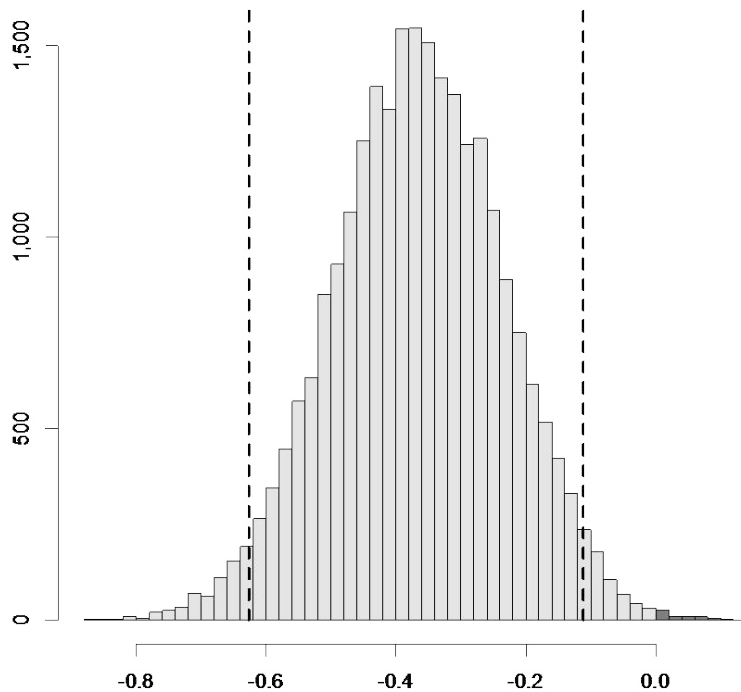
Figure 3. Population effect in fecundity model. Estimated posterior distribution of the mean difference, in logit space, between SRKW and NRKW fecundity rates, 1980–2011. The vertical dashed lines are 95% credible intervals and the probability of a positive difference (SRKW fecundity > NRKW fecundity) is indicated in dark gray.
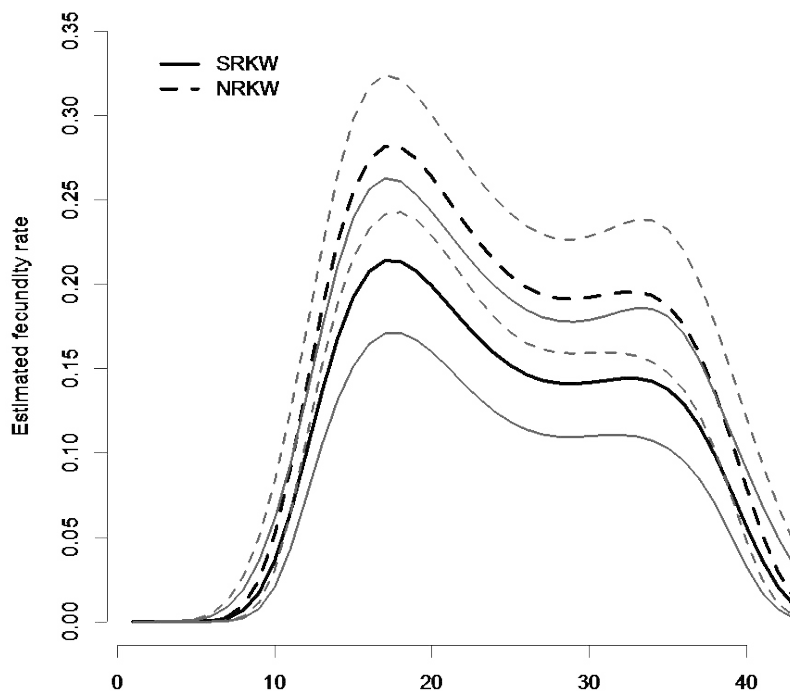
Figure 4. Estimated fecundity rates by age for SRKW and NRKW females, 1980–2011. The thick solid (SRKW) and dashed (NRKW) lines are median estimates as a function of age and the pairs of thinner solid and dashed lines on either side are 95% posterior credible intervals.

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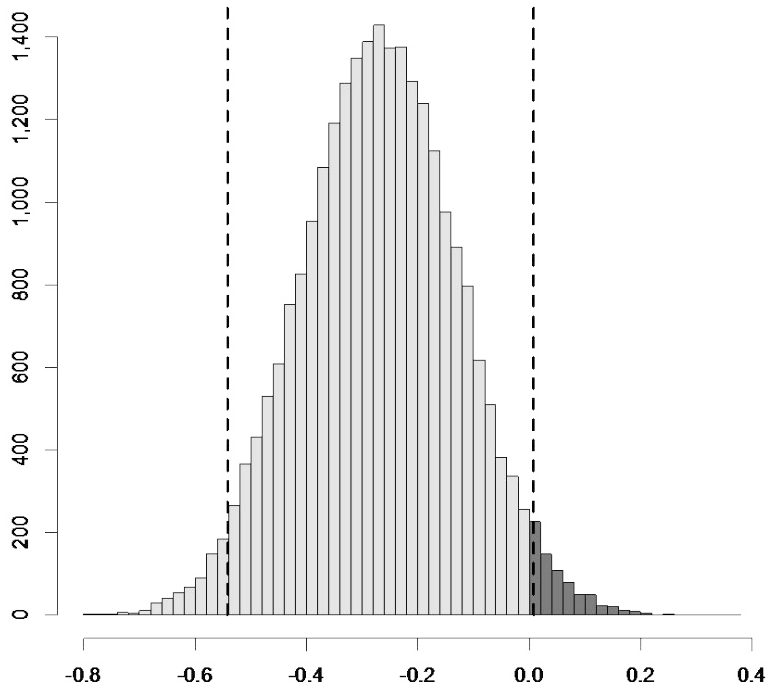
Figure 5. Population effect in survival model. Estimated posterior distribution of the mean difference, in logit space, between SRKW and NRKW survival rates, 1979–2010. The vertical dashed lines are 95% credible intervals and the probability of a positive difference (SRKW survival > NRKW survival) is indicated in dark gray.
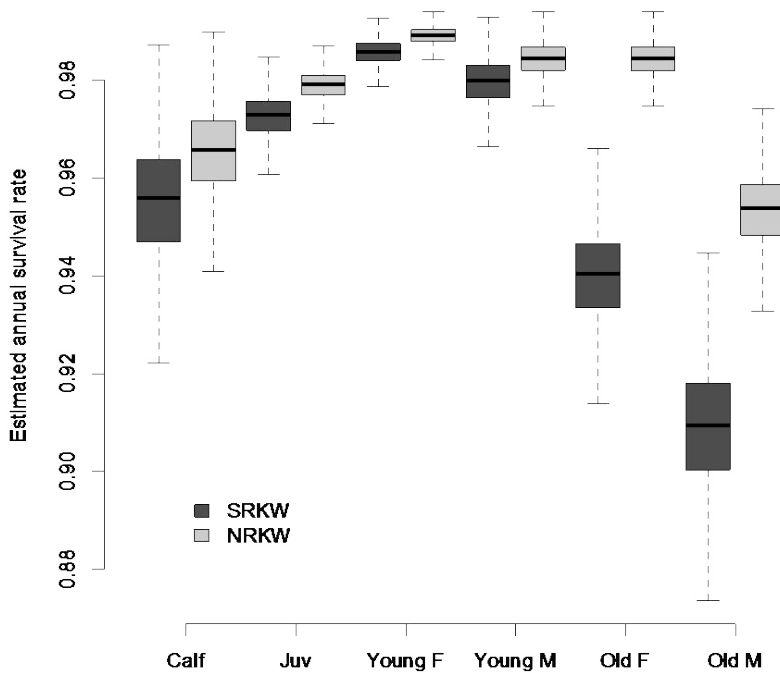
Figure 6. Estimated posterior distributions of survival rates, by stage, for SRKW and NRKW survival rates, 1979–2010. The thick horizontal lines and boxes represent the median and quartiles, respectively. The whiskers indicate the extremes of the data.

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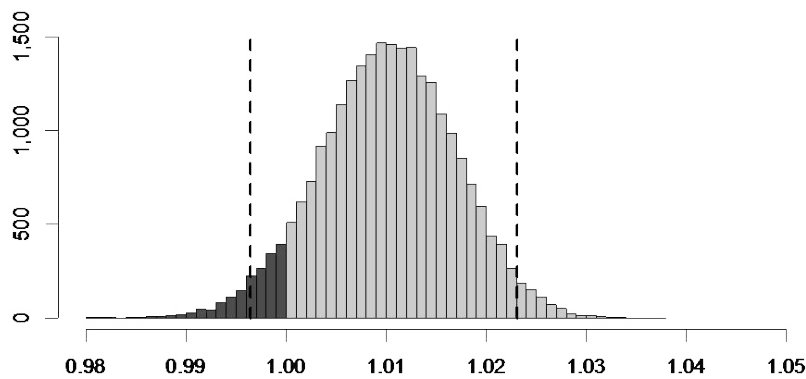
Estimated SRKW (above) and NRKW (below) growth rates
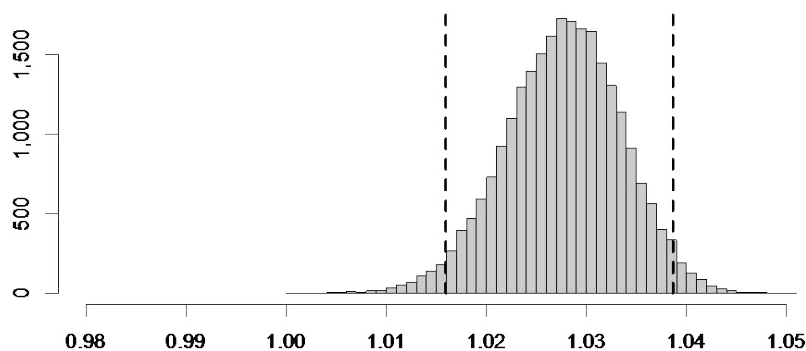
Figure 7. Estimated posterior distributions of growth rates (λ) for SRKW and NRKW populations, 1979–2010. The vertical dashed lines are the 95% posterior credible intervals. For each population, the region of λ corresponding to negative growth (λ < 1) is shown in dark gray.

Table 3. Random effect deviations in fecundity and survival by pod and clan. All deviations are in logit space, but deviations <0 indicate worse than average and those >0 indicate better than average.

Population	Clan	Pod	No. of whales	Fecundity (pod)	Survival (pod)	Fecundity (clan)	Survival (clan)
NRKW	A	A1	22	0.53	0.35	0.11	0.03
	A	A4	16	0.56	−0.28	—	—
	A	A5	13	−0.89	−0.04	—	—
	A	B1	6	0.41	0.07	—	—
	A	C1	17	0.25	0.09	—	—
	A	D1	12	−0.58	−0.18	—	—
	A	H1	5	−0.07	−0.21	—	—
	A	I1	18	0.33	0.07	—	—
	A	I2	3	−0.19	−0.08	—	—
	A	I18	24	0.04	0.05	—	—
	G	G1	34	0.05	−0.06	0.03	0.00
	G	G12	16	0.10	−0.20	—	—
	G	I11	26	−0.12	0.22	—	—
	G	I31	10	0.23	−0.12	—	—
	R	R1	38	0.35	0.36	0.34	0.36
	R	W1	1	0.00	0.17	—	—
SRKW	J	J1	26	−0.15	−0.04	−0.27	−0.15
	J	K1	19	−0.42	−0.04	—	—
	J	L1	39	−0.27	−0.28	—	—

14

Calculating Stock-specific and Aggregate Salmon Indices

Stock-specific Indices

Abundance data were compiled for Chinook salmon stocks from northern British Columbia to the Sacramento River in California to create ocean catch and escapement (OCE) indices. This included terminal run size for fish of both hatchery and natural origin and impacts in ocean fisheries. Data compilation focused on the Pacific Salmon Commission's Chinook Technical Committee's (CTC) Chinook model. The CTC model was selected as a data source because it provides continuous coverage from 1979 through 2010, includes both hatchery and natural production for stocks from Southeast Alaska to the central Oregon coast, and gets around gaps in coverage of coded wire tag (CWT) programs. Some smoothing of the terminal run data is done by the CTC model. For Canadian stocks, the model only has eight stocks, some of which aggregate over different life history types and exploitation patterns, and only four CWT indicator stocks during the model base period. For these reasons, Canadian stocks were disaggregated to a finer scale than the CTC model stocks and 10 CWT indicator stocks were used, where possible, to account for the impacts of ocean fisheries. For stocks not included in the CTC model, outside sources of data were used where available. These include upper Columbia River spring Chinook salmon, Klamath River fall Chinook, and Sacramento River fall Chinook.

These summaries are not exhaustive. All Canadian stock aggregates except Fraser late do not include all production, but they are suitable for describing abundance trends. Coastal Chinook stocks from the Umpqua River south; as well as spring Chinook from the Klamath River; spring, late-fall, and winter Chinook from the Sacramento River; and fall Chinook from the San Joaquin system are also omitted. In all cases, terminal runs and fishery impacts are of fish of ages 3 and older for ocean-type stocks, and ages 4 and older for stream-type stocks, and are presented on a calendar year basis. The abundance estimates (terminal run + fishery impacts) do not account for fish that do not mature, but remain at sea, and they also do not account for predation (e.g., by marine mammals), so they are minimum estimates. When killer whales are at inside waters, the terminal run size of stocks is most representative of the availability of large Chinook salmon. When they are at outside marine waters, the sum of fishery impacts and terminal run is probably more representative.

Comparing the averages from the first 10 years of the time series to those of the last 10 years, there has been a modest decline in overall abundance of Chinook salmon coast wide, but an increase in the terminal run size as a result of reductions in ocean salmon fisheries (Table 4). However abundance patterns differ greatly among the stocks. As an example, the four aggregates of Fraser River spring and summer Chinook stocks appear to be important in the diets of SRKWs (Hanson et al. 2010), and these stocks are collectively incorporated in the CTC model as the Fraser early stock. This model stock includes both stream-type (spring runs 1.2 and 1.3

and summer 1.3) and ocean-type (summer 0.3) life histories, which have very different patterns of distribution, exploitation, and maturation. Stream-type stocks experience little impact from ocean fisheries and are harvested primarily in terminal fisheries, thus their availability to resident killer whales is not substantially impacted by ocean fisheries. Ocean-type stocks are harvested primarily in ocean fisheries. While there has been a modest increase in the abundance and availability of stream-type Chinook from the Fraser River, there has been about a 50% increase in abundance and more than a 300% increase in terminal run size of ocean-type Chinook. This has resulted in more than doubling of the terminal run size of Fraser River Chinook salmon migrating through the inside marine waters during the spring and summer. For other stocks coast wide, increases in abundance have been enhanced and declines in abundance have been mitigated or reversed in terminal runs by reductions in ocean fisheries (Figure 8, Figure 9, and Figure 10).

Aggregate Indices

CTC Indices

CTC abundance indices are intended to provide indices of the abundance of Chinook salmon vulnerable to fisheries over fairly large areas (e.g., Southeast Alaska, northern British Columbia, west coast Vancouver Island, etc.). This is done by using CWT recoveries by stock and age from the model base period (fishing years 1979–1982) to infer the proportion of each stock taken at each age by fisheries during the base period, and expanding these proportions by the age-specific abundance of each model stock during the base period. These catches are summed over stocks and age-classes to provide an estimate of the total catch. In subsequent years, estimates of age-specific abundance, combined with average base-period exploitation rates, are used to produce estimates of what the age-specific catches would have been in the base period with the stock abundances of that year. The abundance index value for that fishery in a particular year is the sum of the scaled catches from that year divided by the average catches in the base period. Thus the abundance indices in any year are intended to reflect what the relative change from the base period catches would have been in that year, if fisheries were conducted as they were during the base period.

FRAM Indices

The Fishery Regulation Assessment Model (FRAM) Chinook model is a single-pool deterministic model of 38 stocks from southern British Columbia to the Sacramento River, excluding some Columbia River spring-run stocks and stocks from southern Oregon including the Klamath River. All marine area fisheries within three time strata (October-April, May-June, July-September) from Southeast Alaska to southern California are included, with finer scale resolution for fisheries within Puget Sound. The number of Chinook salmon by landed catch and total fishery-related mortality is estimated by stock, age, and time period for each fishery. Natural mortality is applied as a constant for each age to each cohort at the start of the time period and does not vary across years, stock, or seasons. Unlike the CWT and CTC approaches, FRAM contains a growth function (vonBertalanffy curve) for mature and immature run types that allows for a size selectivity component to the estimate of Chinook prey for SRKWs. The biological opinion for the Puget Sound Resource Management Plan (NMFS 2011a) used the growth function feature to produce estimates of kilocalories of food energy from larger size

Chinook prey, which were predominant in SRKW scat samples. Because FRAM is a single-pool model, abundance of prey available within a specific spatial location cannot be quantified. However, estimates of Chinook prey abundance were made for coastal and inland waters aggregates within the SRKW range based on historical catch distribution and stock origin region (inland for catches and stocks originating within Puget Sound, Strait of Juan de Fuca, Georgia Strait, and Johnstone Strait, or coastal for catches and FRAM stocks other than inland type).

Whereas the estimates of Chinook prey abundance for the biological opinion of the Puget Sound Resource Management Plan were based on kilocalories of food energy of Chinook salmon that met size selectivity criteria under the provisions of the plan, the indices of Chinook abundance from FRAM for this analysis were not adjusted for size selectivity, nor were the fisheries modified from observed to meet the provisions of the plan. Estimating Chinook prey in terms of kilocalories of food energy was discontinued because of uncertainty in measuring kilocalories of food energy in Chinook tissue, sensitivity of size selectivity parameters in growth functions in FRAM, and the incompatibility of different measurement units between FRAM and the CTC Chinook model. The indices for this analysis were all adult Chinook salmon of age-class 3 to age-class 5 estimated to be in inland waters during July through September of 1983 through 2008 from postseason FRAM runs that incorporate observed catches and terminal run sizes by age and stock.

The stocks and fisheries covered in the FRAM and CTC Chinook models contain much of the same stock-age abundance and CWT base data and both use the same constant natural mortality rates. The structure of the two models is somewhat different, but many of the algorithms for catch and mature run accounting are similar between the two models. Despite these similarities, indices from the CTC Chinook model consistently showed a better fit than the FRAM indices and were used for this analysis and for most of those presented at the bilateral scientific workshops.

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Table 4. Average abundance and terminal run size, in 1,000s of adult fish, for Chinook salmon stock aggregates from northern British Columbia south. Abundance is the sum of terminal run size and impacts in ocean fisheries by calendar year. Averages are presented for the first 10 years (1979–1988) and last 10 years (2001–2010) of the time series compiled for this report.

	Average abundance			Average terminal run		
	1979–1988	2001–2010	Change (%)	1979–1988	2001–2010	Change (%)
Northern British Columbia	117.2	141.7	21	79.2	100.5	27
Central British Columbia	46.6	68.2	46	31.6	46.7	48
West coast Vancouver Island	339.5	220.0	−35	121.8	145.3	19
Upper Georgia Strait	118.2	35.2	−70	30.1	20.8	−31
Georgia Strait hatchery & natural	93.6	65.9	−30	20.2	36.0	78
Fraser late	425.7	207.2	−51	112.6	154.9	38
Fraser spring 1.2	11.2	22.8	103	9.4	21.3	128
Fraser spring 1.3	51.1	50.3	−2	40.9	38.4	−6
Fraser summer 1.3	48.5	53.6	11	34.6	44.8	30
Fraser summer 0.3	149.0	226.0	52	30.4	128.7	324
Puget Sound	641.1	396.9	−38	236.7	237.6	0
Washington coast	156.9	119.0	−24	73.2	72.0	−2
Lower Columbia River spring	130.0	118.8	−9	101.8	102.5	1
Upper Columbia River spring	80.9	222.8	176	80.9	222.8	176
Columbia River summer	36.9	94.9	157	19.2	64.4	236
Columbia River bright fall	369.0	495.0	34	231.0	371.3	61
Columbia River tule fall	461.4	290.9	−37	213.5	195.0	−9
Northern Oregon coast	252.0	188.4	−25	126.3	126.6	0
Klamath River fall	235.3	139.2	−41	100.9	114.0	13
Sacramento River fall	858.0	704.3	−18	212.5	361.7	70
Coast-wide total:	4,622.2	3,861.1	−16	1,906.6	2,605.3	37

18
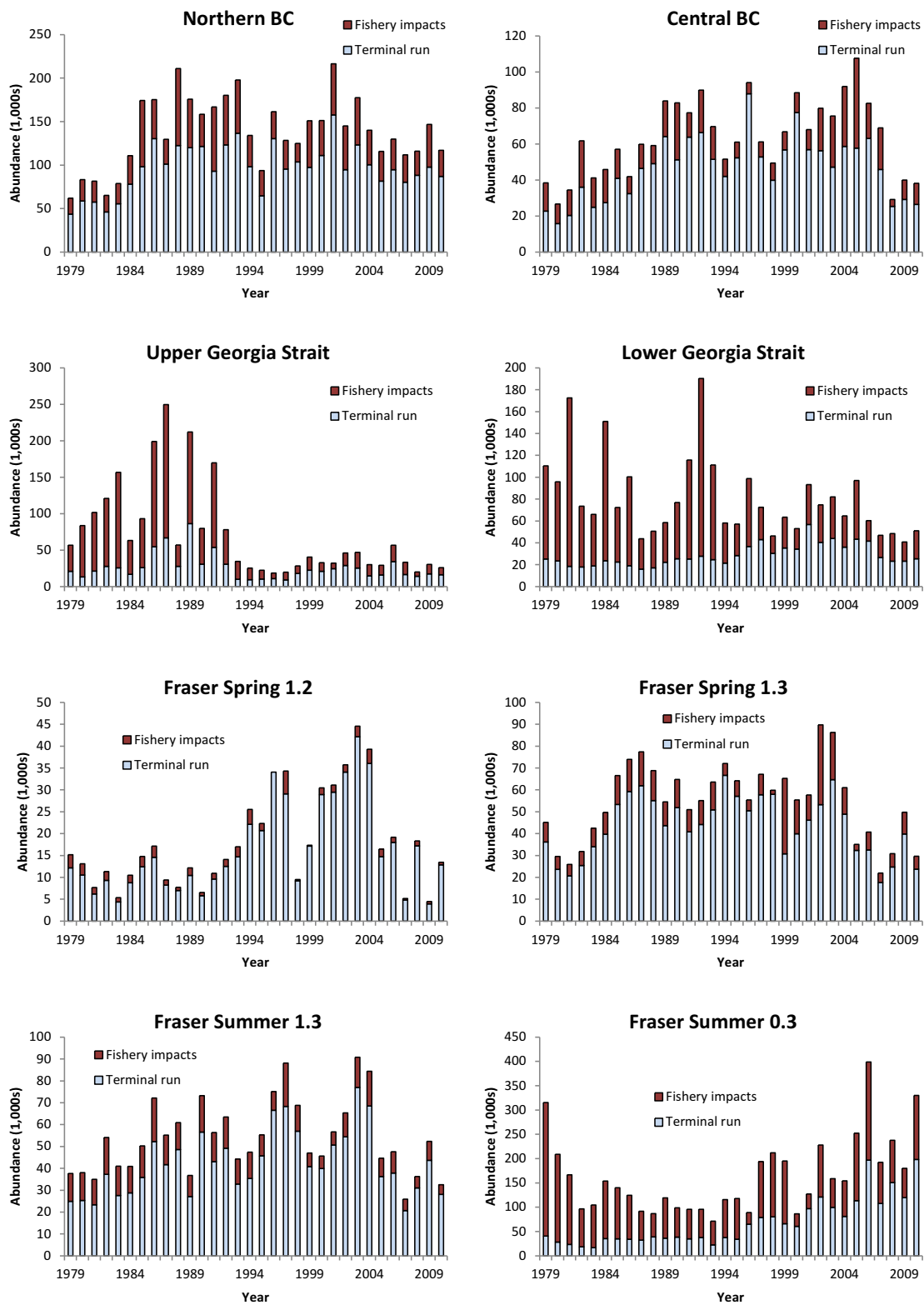
Figure 8. Ocean catch (fishery impact) and escapement (terminal run) indices for eight stocks, 1979–2010.

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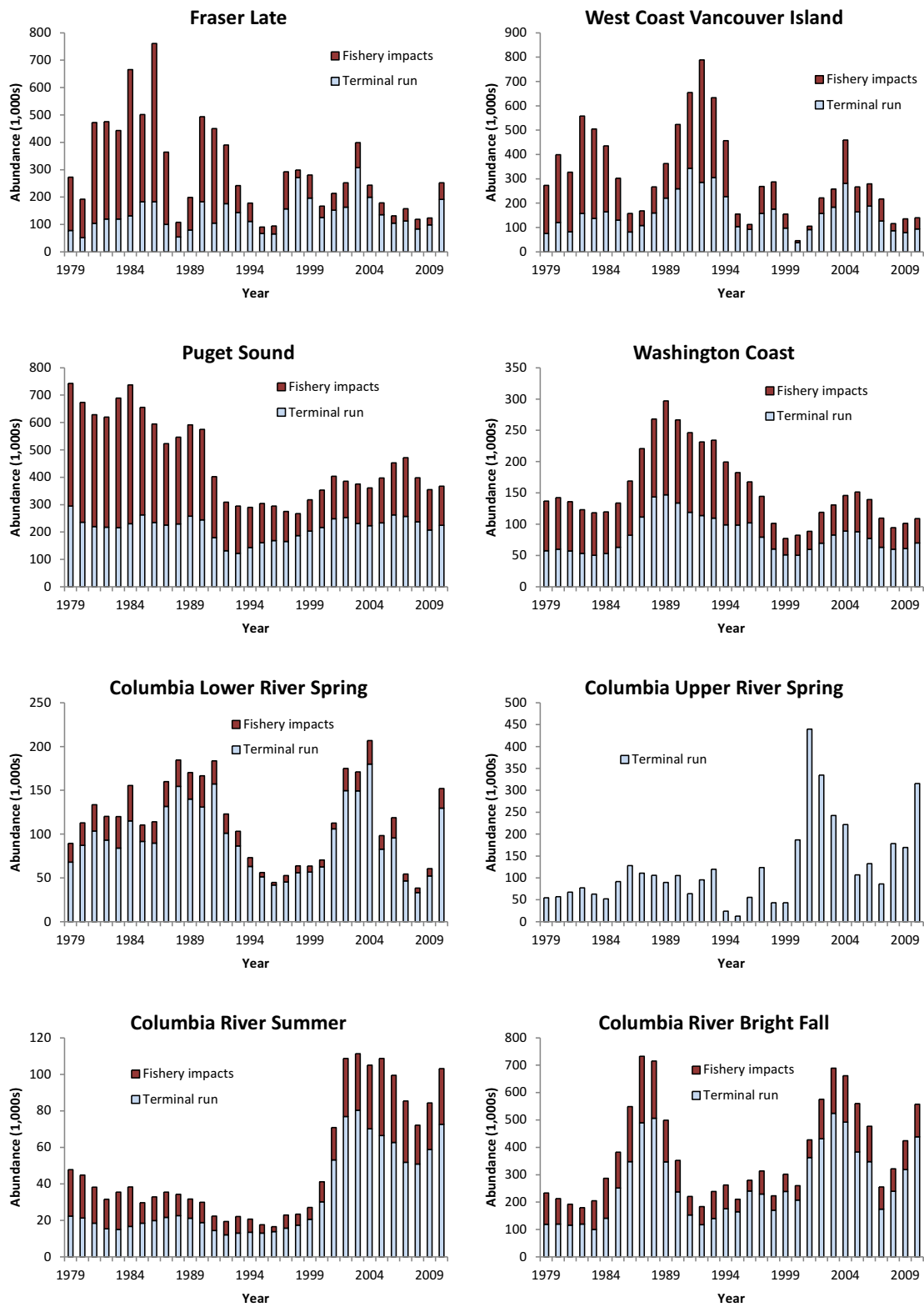
Figure 9. Ocean catch (fishery impact) and escapement (terminal run) indices for eight stocks, 1979–2010.

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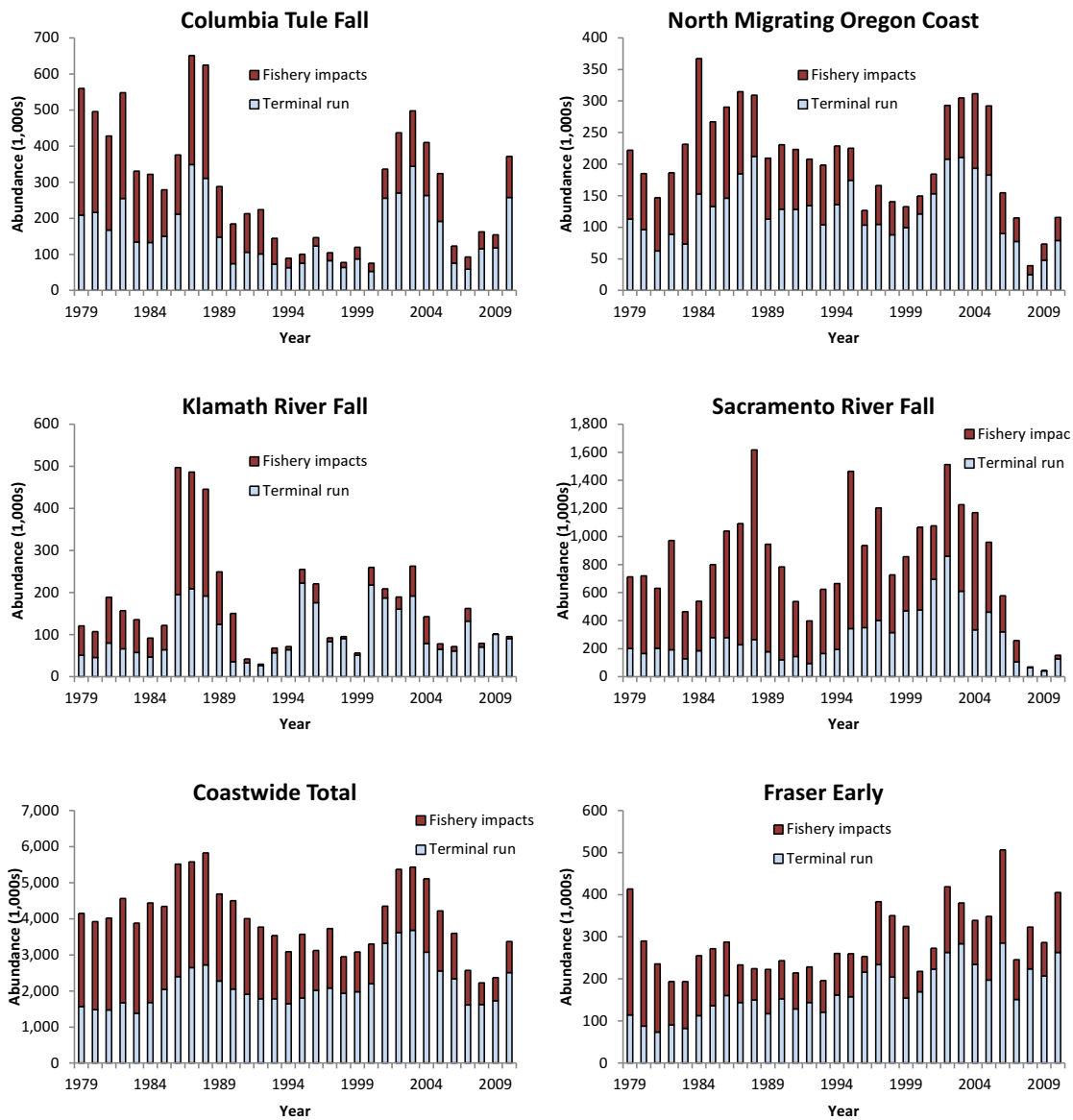
Figure 10. Ocean catch (fishery impact) and escapement (terminal run) indices for six stocks, 1979–2010.

21

Estimating Carrying Capacity and Strength of Density Dependence in SRKWs

Over the period 1979–2010, the SRKW population fluctuated between 71 and 97 individuals, experiencing periods of positive growth and periods of decline, such as in the mid-1990s (Table 2) (NMFS 2008). One of the recurring questions brought up before and during the bilateral workshops was whether one explanation for the small SRKW population growth rate is that the population was at or nearing carrying capacity. We conducted a series of analyses, using SRKW data alone and SRKW data combined with NRKW data to evaluate the strength of evidence for these hypotheses about density dependence and carrying capacity.

The simplest approach to estimating the carrying capacity of the SRKW population is to fit a single-stage population model to total population counts, ignoring age and sex information (Table 2). Two of the most common choices of models are the logistic (or Schaefer) model or the Gompertz growth model. The logistic model is parameterized in terms of the growth rate (r) and carrying capacity (K),

$$N_{t+1} = N_t + rN_t \left(1 - \frac{N_t}{K}\right) + e_t \tag{3}$$

and the Gompertz model can be also expressed in terms of growth (u) and carrying capacity (K),

$$\log(N_{t+1}) = \left(\frac{K-u}{K}\right)\log(N_t) + u + e_t \tag{4}$$

Both models include temporal deviations, e_t, making these models stochastic (process error models) (Hilborn and Mangel 1997). We assume these deviations are distributed $Normal(0, \sigma)$. Unfortunately, carrying capacity estimates from both models are imprecise. The logistic model estimates $K = 88.25$, but the 95% confidence intervals include the majority of observed SRKW population sizes, in addition to an upper 95% interval of 112.4 (Figure 11). Estimates from the Gompertz model are even more uncertain. The inability of these simple models to estimate carrying capacity are not terribly surprising, as SRKWs have fluctuated in a narrow range and have not been observed across extremes of population sizes, such as the ranges observed in pinniped populations (e.g., Jeffries et al. 2003).

Instead of focusing estimation on SRKW carrying capacity, an alternative approach is to estimate the strength of density dependence in SRKWs. Mechanisms of density dependence in cetaceans are poorly understood, but are likely nonlinear and may affect multiple demographic rates (Taylor and DeMaster 2006). Because of the low variation in birth and death rates, we explored whether either observed births or deaths are affected by specific components of population size, such as the total number of mature animals or total number of adult females. Our a priori expectations were that, for both birth rates and survival rates, we would expect density dependent processes to have negative effects (or possibly zero effect). One possible exception is that older, potentially more experienced females, may have offspring with higher

survival rates (Ward et al. 2009b). Because this estimated effect is small and uncertain, we did not include it in our models.

We modified our fecundity and survival models to include effects of prey (indices of Chinook salmon) and proxies for killer whale population density. The modified model became

$$\text{logit}(f_{i,t}) = B_{0,pop} + B_1 \text{age}_{i,t} + B_2 \text{age}_{i,t}^2 + B_3 \text{age}_{i,t}^3 + B_4 \text{age}_{i,t}^4 + B_5 S_{i,t-1} + B_6 X_{i,t-1} \quad (5)$$

where $f_{i,t}$ represents the probability of animal i giving birth at time t, the coefficient $B_{0,pop}$ is an intercept allowing SRKWs and NRKWs to have different mean fecundity rates, B_1, B_2, B_3, B_4 relate the effect of age on the probability of giving birth, the coefficient B_5 relates 1-year lagged Chinook salmon abundance $S_{i,t-1}$ to fecundity (Ford and Ellis 2005), and the coefficient B_6 represents the effect of density dependence on birth rates through some covariate $X_{i,t-1}$. The observed binary data is then modeled as $Y_{i,t} \sim Bernoulli(f_{i,t})$. It is important to note that covariates affect birth rates linearly in link space, but the relationship becomes nonlinear in normal space via the inverse link function. Similarly, we modified our logistic survival model to include covariates,

$$\text{logit}(\phi_{i,t}) = B_{0,pop} + B_{1,\text{stage}} + B_2 L_{i,t} + B_3 S_{i,t} + B_4 X_{i,t} \quad (6)$$

where B_3 and B_4 relate salmon abundance and measures of population size to survival.

In addition to each of the fixed-effects covariates (population, salmon, density), we considered interactions between each of these covariates. Interactions between population (SRKW, NRKW) and Chinook salmon indices were included to allow the NRKW and SRKW populations to have different responses to salmon. The interaction between population and density was included to allow for different strengths of density dependence in each killer whale population. The interaction between Chinook salmon and killer whale density was included to allow the relationship of fecundity or survival and the abundance of Chinook to have a interaction on density that was not simply linear, and finally the three-way interaction between population, killer whale density, and Chinook salmon was included to allow the strength of the salmon–killer whale density relationship to vary by killer whale population.

For both the fecundity and survival models described above, we simultaneously considered CTC indices of Chinook salmon abundance that have been used in previous publications (Ford et al. 2005, Ward et al. 2009a, Ford et al. 2010). Specifically, we focused on northern British Columbia (NBC) and west coast of Vancouver Island (WCVI) indices, obtained from Appendix H in the Pacific Salmon Commission CTC's annual report (Pacific Salmon Commission 2011). These indices were not meant to represent all possible indices of Chinook salmon, but because they are significantly correlated with SRKW demographics, they represent a baseline model to examine effects of density dependence. As proxies for density-dependent metrics, the following population summaries were included as covariates in models of survival and fecundity: total population size, total number of females, total numbers of males, number of adult females, and total number of adult males. Each of these quantities was also tabulated at the pod and matriline levels and assumed unique to each population (females in the NRKW population were not affected by SRKW proxies for density dependence).

We conducted all model selection using maximum likelihood estimation. For all fecundity models considered, we found little support for density-dependent processes affecting birth rates of killer whales (Table 5). The best model in our analysis included an additive population effect and an additive Chinook salmon effect (CTC's NBC index), but no effect of population density (Table 5). Effects of salmon and the difference between killer whale populations were both statistically significant, but all effects of population density were not. The population and salmon effects, without an interaction, indicate that the SRKW population has the same fecundity response to salmon that the NRKW population does, but that NRKW fecundity rates are simply higher (this is consistent with our previous analyses that did not include salmon or density, Figure 4). For the survival models we considered, results were mixed, but appeared to show some weak effects of density dependence (Table 6). The best model included interactions between whale population and Chinook salmon (CTC's NBC index), and unlike the fecundity model, an interaction between these terms was also supported. This interaction can be interpreted as NRKWs and SRKWs having slightly different survival responses to the same change in the Chinook salmon indices. The best survival model also included an interaction between whale population and total females in each population, representing density dependence. This second interaction indicates support for each whale population having slightly different survival responses to similar changes in the total number of females.

The effect size of the estimated interactions on killer whale survival can be calculated by varying the Chinook salmon indices included in the model and the density of adult females across the ranges that have been historically observed. Though we present results for juvenile survival (Figure 12), these results are identical for other life stages in our model, because we treated survival as stage based. What this type of sensitivity analysis suggests is that the NRKW response to density dependence in survival is much stronger than for SRKWs (Figure 12). Part of the challenge in detecting any effects of density dependence in SRKWs is that, over the last 30 years, the SRKW population has exhibited small fluctuations in density when compared to NRKWs. Since 1979, the number of total females in the SRKW population has ranged from 45 to 60, but over the same time period, the number of females in the NRKW population has doubled (from 60 to 120). Because we have not observed SRKWs over such a range of densities, it makes our inference about whether SRKWs are at or near carrying capacity highly imprecise (Figure 11). If density-dependent processes are present, however, we can conclude that their effects on SRKW survival appear to be smaller than variations in other covariates, such as Chinook salmon (Figure 12). Because the effects of density dependence appeared to be relatively small for SRKWs, density-dependent effects were not included in our SRKW population projections.
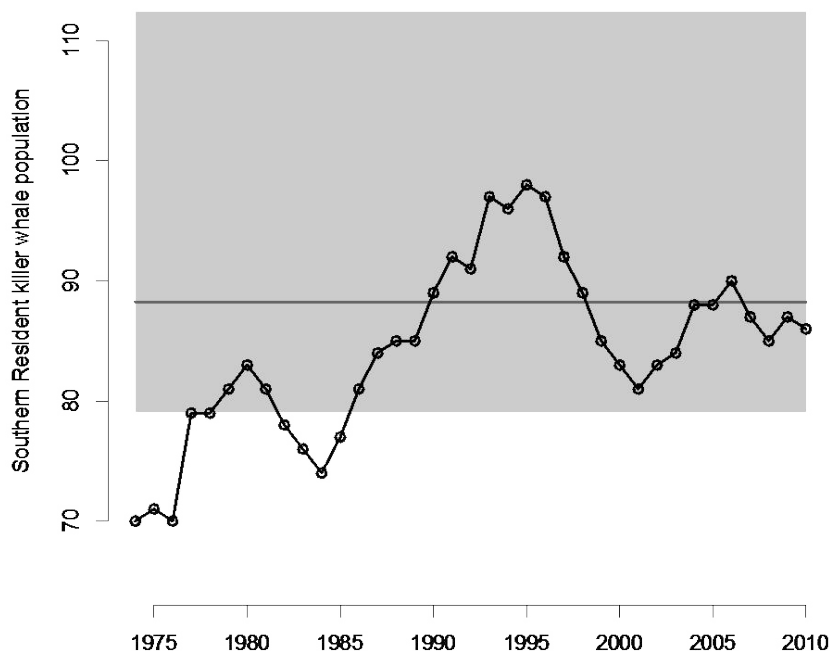
Figure 11. Estimates of SRKW carrying capacity (straight horizontal line) and 95% confidence intervals (light gray rectangular area) from the logistic population growth model, 1974–2010 (Table 5).

Table 5. Fecundity models used to evaluate hypotheses about density dependence and interactions with prey between NRKW and SRKW populations. This table is a subset of the best fitting models. Age is modeled as a fourth order polynomial in each case and the age effect alone represents the null model. Asterisks (*) between terms indicate the inclusion of interaction terms. For all models, two-way interactions also include single fixed effects, and three-way interactions include all pairwise two-way interactions. The best model with the lowest AIC score is in boldface.

Covariates included	AIC	ΔAIC
Null	1918.42	8.89
Population	1911.95	2.43
Population * totalPop	1915.27	5.75
Population * total males	1910.41	0.89
Population * total females	1911.75	2.23
Population * maturePop	1914.71	5.18
Population * total mature females	1914.53	5.01
Population * total mature males	1914.18	4.66
Population * year	1915.77	6.25
Population + CTC WCVI	**1909.52**	**0.00**
Population * CTC WCVI	1911.47	1.94
Population + CTC NBC	1910.62	1.10
Population * CTC NBC	1912.43	2.91
Population * total females * CTC WCVI	1916.50	7.00
Population * total females * CTC NBC	1915.80	6.29
Population * total males * CTC WCVI	1915.00	5.43
Population * total males * CTC NBC	1913.50	4.00
Population * total mature females * CTC WCVI	1915.40	5.90
Population * total mature females * CTC NBC	1916.20	6.71
Population * total mature males * CTC WCVI	1913.70	4.14
Population * total mature males * CTC NBC	1916.30	6.83

25

Table 6. Survival models used to evaluate hypotheses about density dependence and interactions with prey between NRKW and SRKW populations. This table is a subset of the best fitting models. Stage-specific survival is included in every model and the stage effect alone represents the null model. Asterisks (*) between terms indicate the inclusion of interaction terms. For all models, two-way interactions also include single fixed effects, and three-way interactions include all pairwise two-way interactions, in addition to main effects. The best model with the lowest Akaike information criterion (AIC) score is in boldface.

Covariates included	AIC	ΔAIC
Null	2130.0	25.9
Population	2128.4	24.3
Population * totalPop	2118.1	14.0
Population * total males	2126.3	22.2
Population * total females	2111.2	7.1
Population * maturePop	2111.6	7.5
Population * total mature females	2113.5	9.4
Population * total mature males	2111.7	7.6
Population * year	2118.8	14.7
Population + CTC WCVI	2117.8	13.7
Population * CTC WCVI	2119.6	15.5
Population + CTC NBC	2119.3	15.2
Population * CTC NBC	2119.3	15.2
Population * total females * CTC WCVI	2107.5	3.4
Population * total females * CTC NBC	2104.8	0.7
Population * total females + population * CTC NBC	**2104.1**	**0.0**
Population * total females + CTC NBC	2105.0	0.9
Population * total females	2111.2	7.1
Population * CTC NBC	2119.3	15.2
Population * total males * CTC WCVI	2123.0	18.9
Population * total males * CTC NBC	2118.8	14.7
Population * total mature females * CTC WCVI	2116.9	12.8
Population * total mature females * CTC NBC	2105.2	1.1
Population * total mature males * CTC WCVI	2112.3	8.2
Population * total mature males * CTC NBC	2108.6	4.5

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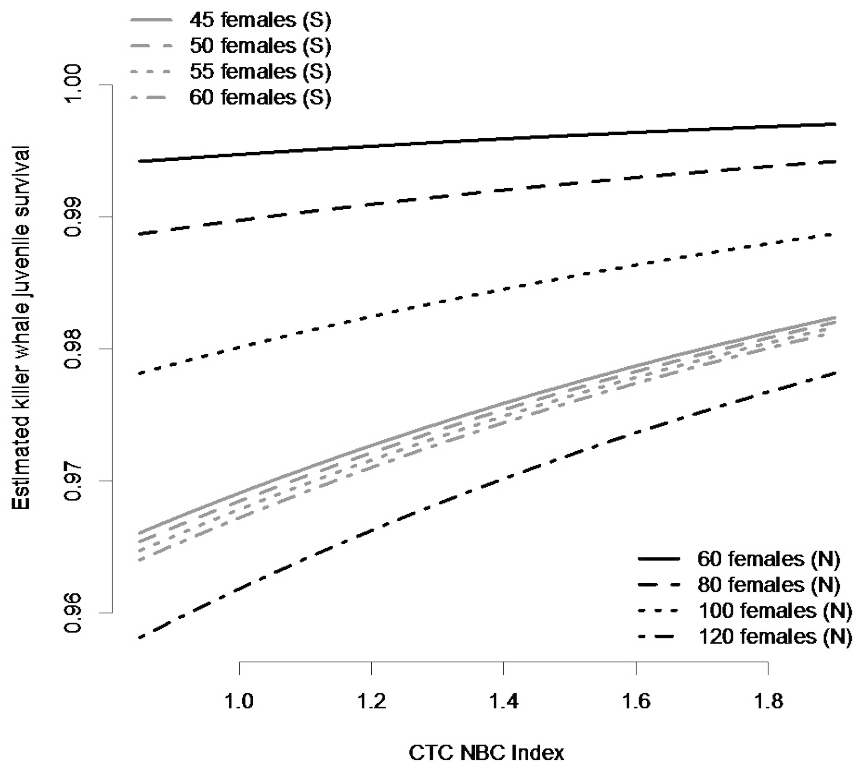
Figure 12. Mean estimates of SRKW and NRKW survival rates as a function of Chinook salmon (CTC northern British Columbia) and respective killer whale population densities. SRKW estimates are shown in gray across the range of total females that have been observed (45–60, corresponding to the tight cluster of four gray lines) and the NRKW estimates are shown in black across the range of total females that have been observed (60–120).

27

Identifying Salmon Stocks Most Correlated with SRKW Population Growth

Previous research has shown that Chinook salmon indices calculated by the Pacific Salmon Commission CTC were correlated with SRKW survival and fecundity (Ford et al. 2005, Ward et al. 2009a, Ford et al. 2010). In work done for the 2011 Puget Sound Chinook Biological Opinion, correlations between SRKW growth and different Chinook indices derived from the FRAM Chinook abundance model were also found (NMFS 2011a). Interpreting comparisons between these two analyses was somewhat confusing, however, due to differences in which years were included in the analyses, differences in salmon stock distribution, differences in salmon age structure (specifically, how juveniles contributed to the indices), and differences in the Chinook stocks included in the indices. In an attempt to resolve any discrepancies between models that included these indices, we focused on how the newer stock-specific OCE indices correlated with SRKW demography. These stock-specific OCE indices were calculated for 21 stock groupings (the Calculating Stock-specific and Aggregate Salmon Indices section above). It is important to note that in comparing the OCE, CTC, and FRAM indices, the three indices rely heavily on CWT recoveries, and as expected, are correlated with each other.

To examine hypotheses about whether SRKW demographics could be correlated with regional or temporal groups of Chinook stocks, we used the 21 OCE indices. One advantage offered by these newer indices is that they include some stocks not included in the CTC indices (Columbia River spring, Sacramento River, and Klamath River Chinook salmon). A second advantage is that they are generated on a stock-by-stock basis, and at finer spatial scale than either the CTC or FRAM indices. There are several weaknesses in this type of modeling approach. First, the models aren't mechanistic, so stocks that are highly correlated with SRKW dynamics are not necessarily the most important or limiting. Second, there is a high degree of correlation between individual stocks and stock aggregates; these correlations in the predictor variables increase the likelihood of spurious correlations and error in the relative ranking of competing models.

We first divided each of the 21 OCE indices into an ocean catch and terminal component (Table 7). The sum of these components was treated as "ocean index" and the terminal component was interpreted as being an "inland waters index," after ocean fishing occurs. For all stocks, we formulated two hypotheses for grouping stocks. First, we were interested in developing spatial indices related to where Chinook salmon are thought to migrate (Groot and Margolis 1991, Quinn 2005, Tucker et al. 2012). We grouped stocks into the following three areas: far north migrating (e.g., Bering Sea), north/central migrating (e.g., Southeast Alaska), and California (or south migrating). Second, we grouped stocks by the run timing of migration into the following groups: spring migrating (peak in March-May), summer migrating (peak in June-August), summer/fall migrating, and fall migrating (peak in September-November). For stocks

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that had multiple designations (e.g., Puget Sound Chinook may be between summer and fall with respect to run timing, Table 7), we explored both alternative model configurations.

In performing model selection, we used the Akaike information criterion (AIC) (Burnham and Anderson 2002) to identify the models most supported by the data. Each of the models was also run in a Bayesian framework, allowing us to examine the relative magnitude of the effect size and the distribution of the posterior probability for each scenario. As a final comparison, we ran all models with just the SRKW population, then a second time with the SRKW and NRKW populations combined (to allow NRKW survival to be slightly different than SRKW, our logistic regressions added one additional parameter for a population offset).

When SRKW data are analyzed separately, all Chinook salmon groupings perform nearly equally well with respect to fecundity (the best model is the ocean index of the central migrating group, without Fraser spring Chinook included) (Table 8). More separation between models is apparent when survival is the response; the fall grouping of Chinook, without California stocks performs well, as does the north migrating group, without including Fraser spring stocks (Table 8). The best fall group included the two stocks categorized as summer/fall (Puget Sound, Strait of Georgia) (Table 8). When the NRKW and SRKW are included in the same model, these results become stronger, with more support going to the grouping of fall migrating stocks (not including California stocks, and only including inland indices after fishing occurs) (Table 9).

Although the fall group (minus California) is used in the remainder of this report to evaluate the effects of ocean fishing on SRKW, we explored several other configurations of stocks to evaluate additional hypotheses. Specifically, members of the Independent Science Panel were interested in how each of the individual stocks compared to the best grouping, and whether a coast-wide aggregate across all 21 stocks performed better (Hilborn et al. 2012). In this comparison, we found that the coast-wide aggregate across all stocks not only performed better than any individual stock, but the coast-wide indices also performed slightly better than the fall grouping used in the remainder of the workshop presentations (delta AIC reduced by 0.76). Although the coast-wide indices appear to be slightly more correlated with SRKW demographics, it is important to emphasize the uncertainty in model selection; delta AIC values less than 2.0 are interpreted as receiving strong data support and should not be removed from the list of candidate models (Burnham and Anderson 2002). Furthermore, because the fall aggregate represents more than 50% of the total Chinook abundance indices over recent years, these indices are highly correlated with the coast-wide aggregate. Therefore, we expect that the results presented here based on the fall group (minus California) will be very similar to what would be expected using the coast-wide indices.

Table 7. Groupings used to evaluate hypotheses about Chinook salmon stocks. "North" in the table refers to stocks that are the far north migrating stocks. Stocks with an asterisk (*) were found as a group to be most correlated with SRKW fecundity and survival (Table 8 and Table 9).

Stock	Run timing	Ocean distribution
Lower Columbia River spring	Spring	Central
Fraser spring 1.2	Spring	North/central
Fraser spring 1.3	Spring	North/central
Southeast Alaska	Spring	North
Upper Columbia River spring	Spring	Central
Areas 1–5	Summer	North
Areas 6–10	Summer	North
Columbia River summer	Summer	Central
Fraser summer 0.3	Summer	North
Fraser summer 1.3	Summer	North
Puget Sound*	Summer/fall	Central
Upper Georgia Strait*	Summer/fall	North
Columbia River tule*	Fall	Central
Columbia River brights*	Fall	North
Fraser River late*	Fall	Central
Lower Georgia Strait*	Fall	Central
Klamath River	Fall	California
Oregon coast*	Fall	North
Sacramento	Fall	California
Washington coast*	Fall	North
West coast Vancouver Island*	Fall	North

Table 8. Model selection results for the SRKW population, 1979–2010. CTC WCVI = CTC index from the west coast of Vancouver Island, CTC NBC = CTC index from northern British Columbia, and NA = not applicable. The delta AIC values are calculated across all models for fecundity, survival, and total (fecundity and survival combined). Lower delta AIC values indicate more data support and the best model for each comparison has a delta AIC value of 0.

Group	Include inland preterminal?	ΔAIC fecundity	ΔAIC survival	ΔAIC total
No salmon	NA	1.7	15.4	13.6
CTC WCVI	NA	2.5	4.7	3.7
CTC NBC	NA	3.0	0.6	0.0
Distribution hypotheses				
North (+Fraser spring)	Y	1.7	2.4	0.6
North (+Fraser spring)	N	3.5	2.1	2.1
North (−Fraser spring)	Y	3.4	2.1	2.0
North (−Fraser spring)	N	3.5	1.5	1.5
Central (−Fraser spring)	Y	0.0	7.1	3.5
Central (−Fraser spring)	N	3.2	7.8	7.4
Central (+Fraser spring)	Y	2.7	7.4	6.6
Central (+Fraser spring)	N	3.2	7.9	7.6
South	NA	3.6	17.3	17.4
Run timing hypotheses				
Spring	Y	2.6	12.6	11.7
Spring	N	2.8	13.0	12.2
Summer	Y	3.5	17.0	17.0
Summer	N	3.7	16.2	16.4
Summer/fall	Y	2.1	12.6	11.1
Summer/fall	N	1.9	12.3	10.6
Fall (+California)	Y	3.7	4.4	4.5
Fall (+California)	N	3.7	4.5	4.7
Fall (−California)	Y	3.6	1.7	1.7
Fall (−California)	N	3.7	0.0	0.1

31

Table 9. Model selection results for the SRKW and NRKW populations, 1979–2010. CTC WCVI = CTC index from the west coast of Vancouver Island, CTC NBC = CTC index from northern British Columbia, and NA = not applicable. The delta AIC values are calculated across all models for fecundity, survival, and total (fecundity and survival combined). Lower delta AIC values indicate more data support and the best model for each comparison has a delta AIC value of 0.

Group	Include inland preterminal?	ΔAIC fecundity	ΔAIC survival	ΔAIC total
No salmon	NA	2.4	20.6	21.1
CTC WCVI	NA	0.0	10.1	8.2
CTC NBC	NA	1.1	11.5	10.6
Distribution hypotheses				
North (+Fraser spring)	Y	2.4	7.2	7.7
North (+Fraser spring)	N	2.7	9.2	9.9
North (−Fraser spring)	Y	2.1	5.4	5.6
North (−Fraser spring)	N	2.8	7.9	8.7
Central (−Fraser spring)	Y	3.2	18.5	19.8
Central (−Fraser spring)	N	0.9	22.3	21.3
Central (+Fraser spring)	Y	1.6	18.2	17.8
Central (+Fraser spring)	N	1.0	22.2	21.3
South	NA	4.4	16.9	19.4
Run timing hypotheses				
Spring	Y	3.1	15.4	16.5
Spring	N	3.1	15.1	16.3
Summer	Y	0.7	19.2	18.0
Summer	N	3.0	15.1	16.2
Summer/fall	Y	3.5	12.8	14.4
Summer/fall	N	3.8	19.2	21.1
Fall (+California)	Y	2.7	14.6	15.4
Fall (+California)	N	2.6	15.7	16.4
Fall (−California)	Y	2.4	2.2	2.6
Fall (−California)	N	1.9	0.0	0.0

Estimating Historical Effects of Chinook Abundance and Salmon Fishing on SRKW Population Growth

To explore possible historical effects of fishing on SRKW growth rates, we conducted a retrospective analysis, using the indices of fall Chinook stocks (Table 7, Table 8, and Table 9). The benefit of using these indices is that we can partition them into a time series of fisheries impacts and a time series of terminal run size (Table 10). To approximate the past effects of fishing on past SRKW population growth, we calculated the expected value of λ in each year 1979–2010, first using the terminal run only as a predictor (to model a scenario with ocean fishing), then using the total ocean index (catch + escapement) as a predictor. The second scenario, meant to represent a situation with no ocean fishing, is somewhat unrealistic in that it assumes all fish forgone by fishing would be made available to the SRKW population.

Using the logistic modeling procedure described above, we first estimated the posterior distributions of the regression parameters using Bayesian methods, saving 25,000 vectors of MCMC draws. Second, for each vector of MCMC parameters, we generated the expected fecundity rate for each age-class (for all SRKW females of reproductive age, 10–42) and the expected survival rates, by stage, for animals through age 42. Because postreproductive females have no impact on population growth or estimation of λ, they were not included in the analysis. Third, these fecundity and survival rates were used to construct an age-based transition matrix (Fujiwara and Caswell 2002) with survival rates on the subdiagonal. For each reproductive age, we allowed females to be in a "breeding" class or "nonbreeding" class, depending on whether they reproduced. Females that gave birth in year t automatically entered the nonbreeding class the following year, because the 18-month gestation in SRKWs prevents females from giving birth in successive years (Ford and Ellis 2005). Calves were fractionally assigned as females, using a normal approximation to the binomial distribution, and given the historical SRKW data on sex ratio at birth 1979–2010, $\Pr(female) \sim Normal\left(u = \frac{48}{96}, \sigma = \sqrt{\frac{\frac{48}{96} \cdot \left(1 - \frac{48}{96}\right)}{96}}\right)$. Nonbreeding females in year t transitioned to becoming breeders with probabilities equal to the estimated age-specific fecundity rates estimated above $\left(f_{i,t}\right)$. Given the transition matrix, we calculated λ as the dominant eigenvalue. Because this analysis was done in a Bayesian framework, we repeated this analysis for each of the 25,000 MCMC vectors, yielding a posterior probability distribution for λ.

To evaluate the effects of an increase in salmon abundance on SRKW population growth, we calculated the absolute and percent improvement in λ. Absolute improvement was calculated as $\lambda_{t,total} - \lambda_{t,terminal}$, or the difference in expected growth rates without and with a fishing impact. A more positive absolute difference translates into a greater impact of fishing. The percent improvement in λ was calculated as $100 \cdot \lambda_{t,total}/\lambda_{t,terminal}$, where values greater than

100 correspond to a greater effect of fishing. Each of these calculations was done separately for each of the three SRKW pods (J, K, and L) to investigate interactions between the estimated effects of increasing salmon abundance and the age structure of each pod.

Using terminal run alone of the aggregated fall Chinook index shows considerable differences between pods (Figure 13), and that in most years, the expected pod-specific growth rates remain slightly positive. It is important to note that these represent the expected rather than observed growth rates, and this only quantifies the female growth rate. The patterns or trends across years is largely a function of shifting age structure in each pod; for examples, J Pod has recruited more reproductive females than other pods and L Pod has a sex distribution currently skewed toward more males than females. Across the entire time series, 1979–2010, the effect of a 20% increase in Chinook salmon seems to be highest in periods when estimated harvest rates on salmon were high (e.g., before 1985, Table 10) or in years with low salmon abundance (mid-1990s). As a last point, in most years, the 95% posterior credible intervals on the absolute and percent change are wide and even contain negative values (Figure 14 and Figure 15). This uncertainty is a result of demographic stochasticity, due to the SRKW population being small.

Table 10. Time series of terminal run, fishery impact, and combined total ocean index for the fall grouping of Chinook salmon (without California stocks, Table 7, Table 8, and Table 9). Numbers are in 1,000s of fish.

Year	Terminal run	Fishery impact	Total ocean index	Year	Terminal run	Fishery impact	Total ocean index
1979	993.0	1500.8	2493.8	1995	883.1	420.0	1303.1
1980	937.7	1449.7	2387.4	1996	942.3	358.2	1300.5
1981	849.8	1680.3	2530.1	1997	1028.4	545.1	1573.5
1982	1057.1	1768.8	2825.9	1998	1065.3	352.6	1417.9
1983	875.5	1766.1	2641.6	1999	1030.0	395.8	1425.8
1984	1045.9	1956.1	3002.0	2000	867.7	299.1	1166.8
1985	1223.3	1332.4	2555.7	2001	1402.9	417.8	1820.7
1986	1360.3	1634.8	2995.1	2002	1623.1	637.5	2260.6
1987	1650.6	1374.9	3025.5	2003	1952.4	667.3	2619.7
1988	1661.5	1076.0	2737.5	2004	1791.8	727.4	2519.2
1989	1422.0	1142.6	2564.6	2005	1437.3	683.6	2120.9
1990	1315.3	1352.7	2668.0	2006	1221.2	524.3	1745.5
1991	1211.8	1415.4	2627.2	2007	913.6	503.9	1417.5
1992	1119.0	1418.2	2537.2	2008	885.5	352.6	1238.1
1993	1032.2	1002.0	2034.2	2009	971.3	362.6	1333.9
1994	986.4	715.9	1702.3	2010	1396.3	475.3	1871.6
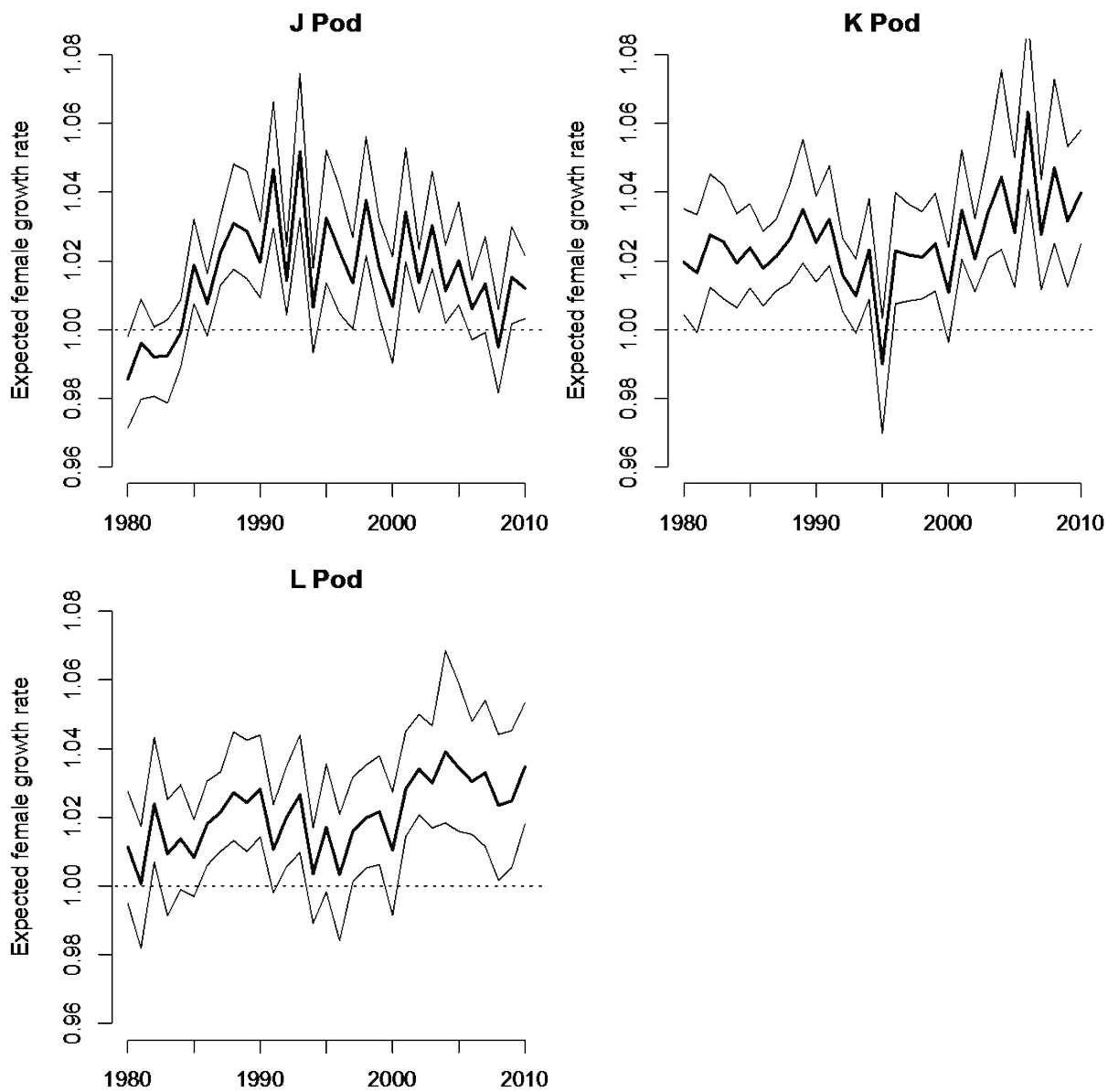
Figure 13. Estimates of pod-specific SRKW female growth rates, using just the terminal run of the fall group of Chinook as a predictor (Table 10). The thick line in each graph is the median growth rate for each pod and the thin lines are 95% posterior credible intervals. Horizontal dashed lines indicate no growth.
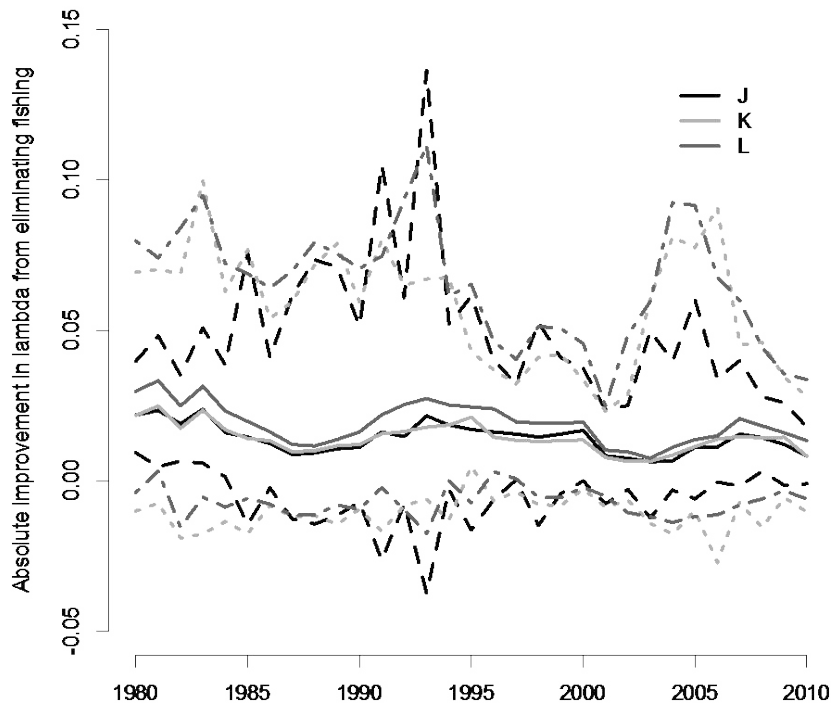
Figure 14. Absolute improvement in SRKW female growth rate by year, calculated by comparing the fall Chinook salmon index (Table 10) with and without the fishing impact. The three solid lines are the median estimates for each pod and the dashed lines are 95% posterior credible intervals.
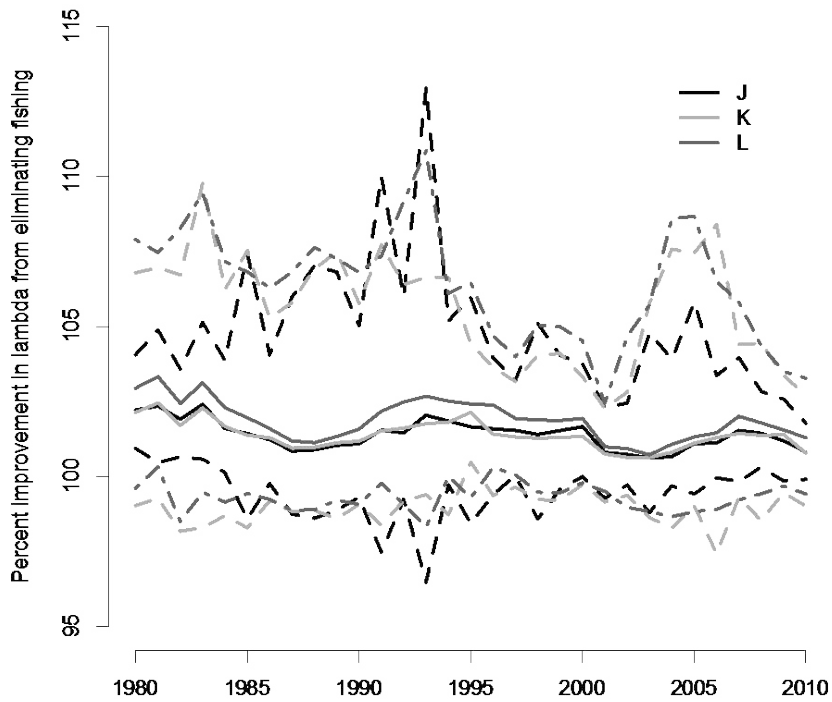
Figure 15. Percent improvement in SRKW female growth rate by year, calculated by comparing the fall Chinook salmon index (Table 10) with and without the fishing impact. The three solid lines are the median estimates for each pod and the dashed lines are 95% posterior credible intervals.

Estimating Future Effects of Changing Chinook Salmon Abundance on SRKW Population Growth

In addition to performing the retrospective analysis, we were interested in estimating how changes in average Chinook abundance in the future are expected to influence the future growth rate of the SRKWs. These analysis were motivated by the question of whether an increase in Chinook abundance due to reduced fishing would have a substantial effect on SRKW recovery trends, but the analysis does not explicitly assume any particular cause to the change in Chinook abundance and obviously such changes could arise from any number of factors.

Our historical analysis of SRKW growth rates was constrained to only including females, because of interest in time-varying calculations of the growth rate (Hilborn et al. 2012). A second limitation was that our historical analysis did not include all sources of stochasticity, because at each year, the SRKW population size and age and sex structure was known without error. Chance events, such as those generated by demographic stochasticity, are especially important for small populations like the SRKW population. As a more complete analysis, we wanted to perform future projections of the entire SRKW population, including all sources of uncertainty. This type of projection or population viability analysis allows us to evaluate not only the possible effects of increasing Chinook salmon abundance (similar to reducing fishing levels), but also allows us to explore the effects of reaching recovery thresholds that would allow the SRKW population to be downlisted or delisted from ESA.

We investigated several management criteria and benchmarks for SRKW recovery. NMFS established a quantitative downlisting (from endangered to threatened) criterion of SRKWs maintaining a growth rate of 2.3% per year for 14 years (NMFS 2008). Similarly, part of the NMFS delisting criteria for the SRKWs is a maintained growth rate of 2.3% for 28 years. The 2.3% growth rate in both instances represents a growth rate that the SRKW population sustained for 12 years (1984–1996) prior to ESA listing (Krahn et al. 2002, Krahn et al. 2004) and that the NRKW has sustained over a period of several decades. As an additional recovery metric, we also evaluated the likelihood of Puget Sound Partnership's (PSP) goal of reaching a SRKW population size of 95 whales by the year 2020 (PSP 2009). Though the science panel was also interested in probabilities of extinction or quasi-extinction (Hilborn et al. 2012), we focused on the aforementioned criteria because, with a slightly positive growth rate, it was difficult for our simulations to drive the SRKW population to extinction.

Simulations were conducted with a time horizon of 30 years into the future. This time horizon represents a balance between a desire to evaluate the effects of alternative management scenarios on long term outcomes (e.g., the delisting criteria require a minimum of 28 years of information) and the recognition that the validity of the simulation results becomes more uncertain the further they are projected into the future. Uncertainty in the future projections arises not only because of the demographic uncertainty inherent in a small population (and

accounted for in the projections), but also because the key assumption of stationarity (that the model estimated from past data will remain valid in the future) becomes more tenuous the further it is projected into the future.

The first step in our population projections involved simulating time series of future salmon abundance, generated by a similar stochastic process that could have produced the historical estimates (Table 10). Using levels of autocorrelation and a process coefficient of variation that is similar to that in our index of fall Chinook salmon (lag-1 autocorrelation = 0.7, $\sigma = 0.2$), we simulated time series that were as long as each of our SRKW projections (30 years). Each simulated Chinook time series was detrended to force a constraint of stationarity, and to initially have a mean equal to that of the observed time series (Table 10). We then considered a range of hypothetical decreases or increases in the fall Chinook OCE index, ranging from 600,000 to 1.7 million Chinook salmon in steps of 100,000.

Initializing the simulated whale populations from the SRKW population at the end of 2011, we created an individual-based model from our estimated posterior distributions of 1) fecundity rates, 2) survival rates, and 3) distribution of sex ratio at birth. In modeling the dynamics of small populations, individual-based models can better capture uncertainty resulting from demographic stochasticity compared to other approaches. By treating each of the demographic parameters as a probability distribution rather than a single value, we are propagating our parameter uncertainty forward in these projections. Each realization of our projection proceeded as follows:

1. A random time series representing a salmon terminal run size was generated (starting with the lowest abundance, 0.6 million).
2. Unsexed SRKW individuals in 2011 were randomly assigned a sex, assuming sex ratio at birth is $\Pr(female) \sim Normal\left(u = \frac{48}{96}, \sigma = \sqrt{\frac{\frac{48}{96}\cdot\left(1-\frac{48}{96}\right)}{96}}\right)$.
3. Females available to give birth were each assigned a randomly drawn MCMC vector, used to generate fecundity rates.
4. Calves were randomly assigned to these females, with sex assigned similarly to step 2, treating each as a Bernoulli draw.
5. All individuals in the SRKW population were assigned a randomly drawn MCMC vector used to generate survival rates.
6. Bernoulli draws were used to determine which individuals survived to the next time step.
7. Steps 3 through 6 were repeated for the remaining 29 years, representing 30 total time steps.
8. The randomly generated salmon time series was increased by 0.1 million salmon, and steps 3 through 8 repeated.
9. Steps 1 through 8 were repeated 25,000 times.

The matriline, pod, and mother of every hypothetical animal was stored in each iteration, and realistic rules were used to constrain reproduction and survival. First, calves whose mothers died during step 6 were also removed from the population. Second, females were not allowed to give birth in successive years. Third, females were only able to reproduce if a reproductive-aged male (>12 years old) was alive in a different matriline from the female in question. Mating within a pod was allowed, following recent information about paternity (Ford et al. 2011).

For each level of Chinook salmon abundance, we calculated the following quantities on a yearly basis across all simulations: 1) mean and variance of the SRKW population size, both in total, and by pod; 2) the change in SRKW population size, in total and by pod, relative to average under the status quo fall Chinook abundance estimate of 1.2 million; 3) the probability of SRKW reaching a total population size of at least 95 animals by 2020 (PSP recovery goal); 4) the probability of negative population growth under different levels of Chinook abundance; 5) the expected change, in whales per year, relative to the status quo terminal run size; 6) the probabilities of meeting the NMFS downlisting and delisting criteria; and 7) the relative change in the probabilities of meeting the recovery criteria.

Our projections indicated that under the status quo (a mean fall terminal run size of 1.2 million Chinook salmon), the SRKW population as a whole is expected to continue to grow slowly (Figure 16, Appendix B). However, L Pod is expected to decline slightly, because of an age structure currently skewed toward older individuals and a sex structure biased toward males. Over 30 years, the projections indicate that K Pod is expected to increase by several animals and the majority of the growth is from J Pod (Appendix B). Under all scenarios, the projections become highly uncertain more than several years in the future because of demographic stochasticity (Figure 17, Appendix B). This complicates our ability to quantify effects of management actions. Even modeled scenarios in which the average population size of the SRKWs grows compared to the status quo have a wide range of potential outcomes, including population declines, simply because of high levels of demographic stochasticity (Figure 16).

For recovery goals, increasing Chinook salmon had an expected impact on reducing the chance of negative SRKW growth (Figure 18). Our estimates of achieving the PSP goal of 95 animals in 2020 was 47.7% under the status quo, and as high as 83% if the Chinook index was increased by 40% (Figure 19). In terms of reaching the recovery goals required for delisting and downlisting under ESA, we expect the probability of either goal to be less than 25% over the next 30 years if salmon abundance is maintained close to the status quo (Figure 20). As expected, downlisting is more likely than delisting because the growth rate of 2.3% needs to be maintained for a shorter time interval (14 years vs. 28 years). Uncertainty is not presented for these metrics or the PSP goal, because credible intervals are generated from a binomial distribution and are solely a function of the number of simulations. However, these simulations include the full range of parameter uncertainty in the modeled relationships between salmon and killer whales. Increasing the Chinook salmon index by 20% translates to a simulated increase in the delisting and downlisting probabilities by 15% and 12%, respectively (Figure 21).

Growth rate metrics like the 2.3% downlisting and delisting criteria for SRKW offer meaningful targets for recovery, but any single metric has limitations. An obvious criticism of growth rate metrics is that a population may hypothetically grow at a rate less than necessary to achieve delisting or downlisting, but could nonetheless eventually become large enough to suggest that it is not at risk of extinction. In modeling exercises related to the final workshop, we explored several potential two-dimensional metrics balancing growth rate metrics against measures of risk. In this exercise, we used 30 females (≤42 years old) to represent quasi-extinction, and projected the SRKW population forward 30 years. This metric for quasi-extinction is somewhat arbitrary, but unlike actual extinction (defined as 0 individuals) or previously used quasi-extinction metrics (e.g., 10 males and 10 females, Krahn et al. 2004), the probability of reaching quasi-extinction is non-zero. The benefit of using 30 females to represent

quasi-extinction is that we can also assume it is proportional to other levels of quasi-extinction (10, 20, or 40 females), as well as actual extinction. Because it is consistently non-zero, it is a more informative measure of risk compared to actual extinction (which occurs with a much lower frequency). Given our definition of quasi-extinction, we can plot probabilities of recovery and metrics for quasi-extinction for our 30-year simulations on single plots as a function of Chinook salmon indices (Figure 22).

As a more retrospective example of balancing growth and risk metrics, we calculated short-term annual growth rates against the probability of the SRKW population reaching quasi-extinction. Each of these quasi-extinction probabilities was estimated by projecting the SRKW population forward 30 years from the age structure in year t, then calculating the fraction of simulations that produced trajectories below this threshold. For each year included in our analysis, we also calculated the short-term growth rate over the year before and after each census, $1 + \log(N_{t+1}/N_t)/2$. Plotting the relationship between risk and short-term growth (Figure 23) illustrates that, while growth rates were low in recent years, over the entire time series 1979–2010, the worst period for SRKWs was in the early 1980s. During this period, the SRKW population was at low density, had a particularly skewed age and sex structure (as an immediate effect of the aquarium harvest (Bigg and Woman 1975), and as a result, the SRKW population had both a low expected growth rate and a relatively higher risk of extinction, compared to periods later in the time series (such as the late 1990s).
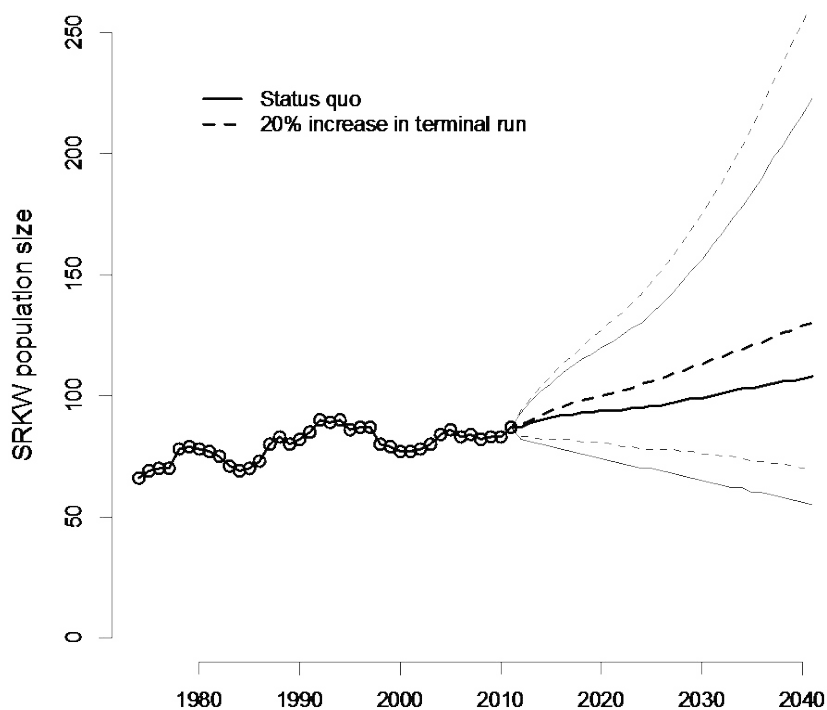
Figure 16. Observed and projected SRKW population size, years 1974–2040. Results from the annual census are indicated with a circle and the thick solid line represents the projected median population size under the status quo level of 1.2 million Chinook salmon (with 95% posterior credible intervals in thin solid lines). The thick dashed line represents the median SRKW population size under an increase of 20% to 1.44 million Chinook salmon (with 95% posterior credible intervals in thin dashed lines).

40
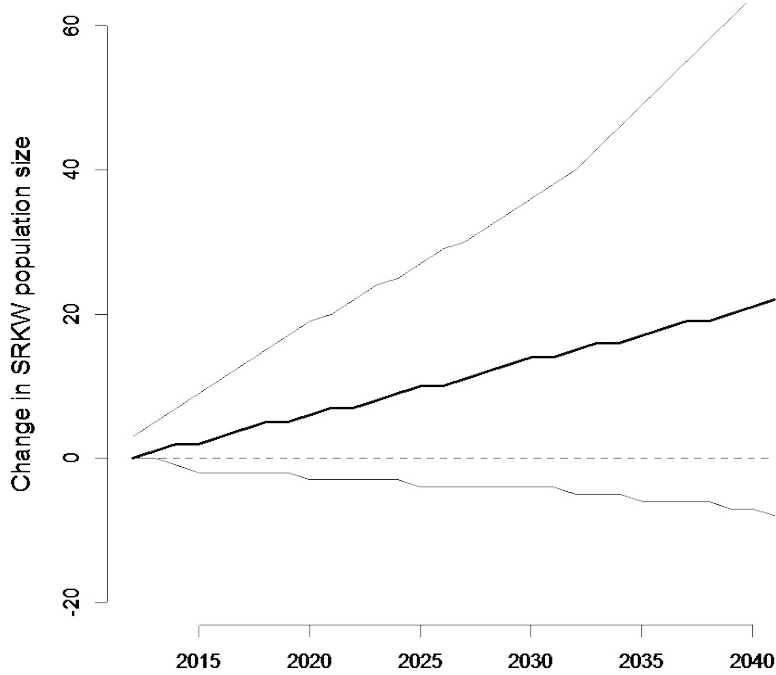
Figure 17. Expected change in SRKW population size by year relative to an increase of Chinook salmon by 20%, from 1.2 to 1.44 million. The expected change is the difference under the two scenarios, with the median indicated by the solid thick line and 95% posterior credible intervals shown as thin solid lines. No change is indicated by the thin dashed line.
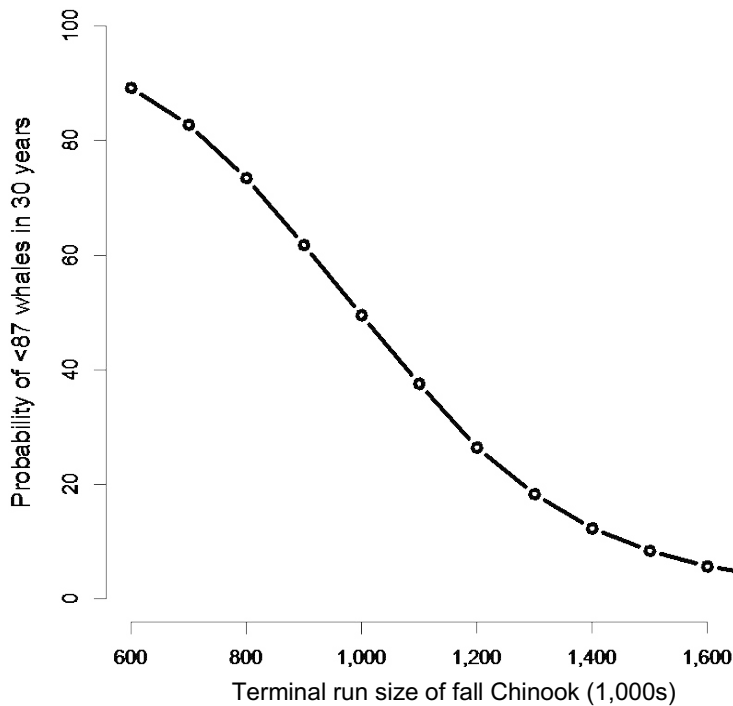
Figure 18. Probability of negative long-term SRKW population growth in 30 years, relative to the 2011 population size of 87 whales, as a function of different levels of Chinook abundance. The status quo is represented as 1.2 million salmon. Estimates are generated from 25,000 simulations.

41
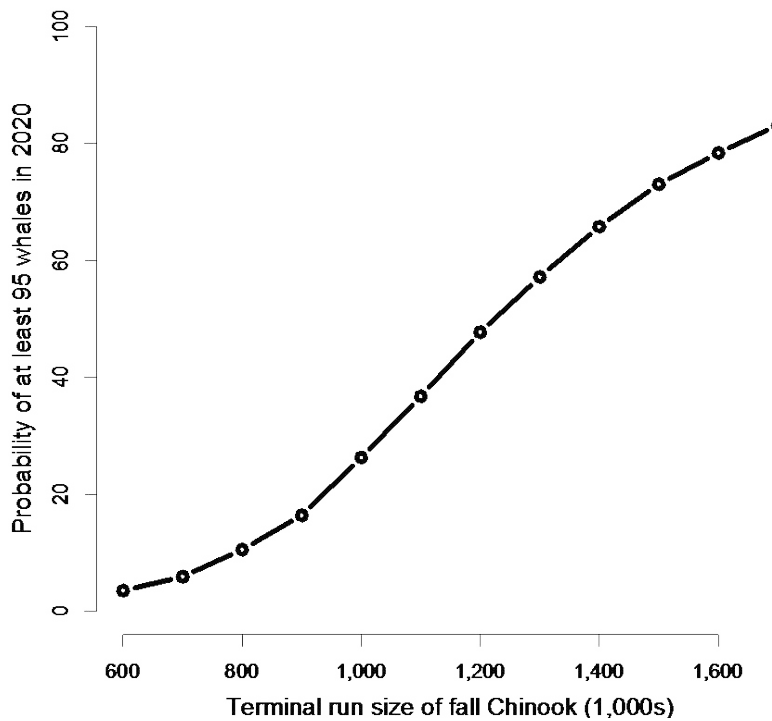
Figure 19. Probability of meeting the PSP recovery goal of at least 95 whales in the SRKW population by the year 2020. The status quo is represented as 1.2 million Chinook salmon. Estimates are generated from 25,000 simulations.
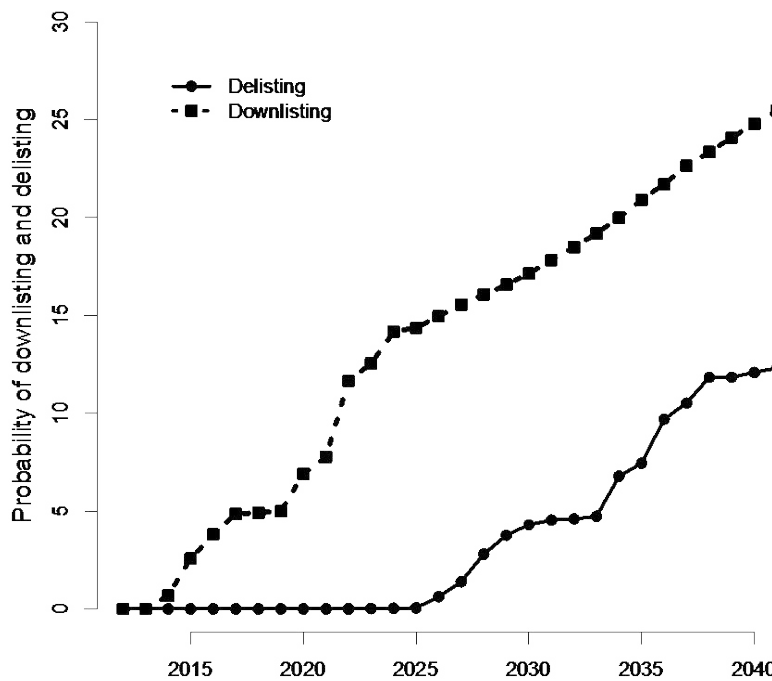
Figure 20. Probability of meeting the recovery goals required for downlisting and delisting the SRKW population under the ESA (2.3% growth for 14 and 28 years, respectively) by year. Estimates are generated from 25,000 simulations, under the status quo scenario (1.2 million Chinook salmon).

42
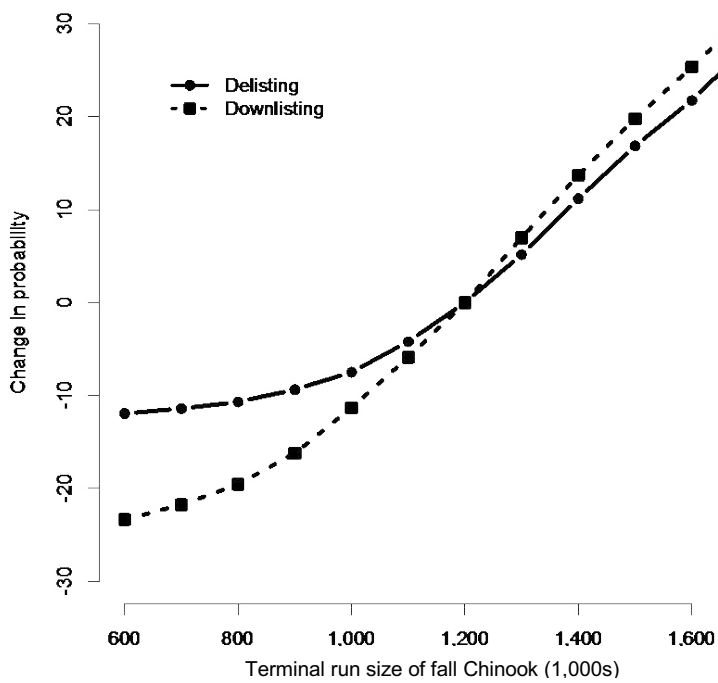
Figure 21. Relative changes in the probabilities of meeting the recovery goals required for downlisting and delisting the SRKW population under ESA (2.3% growth for 14 and 28 years, respectively). Estimates are from 25,000 simulations, and relative to the status quo (1.2 million Chinook).
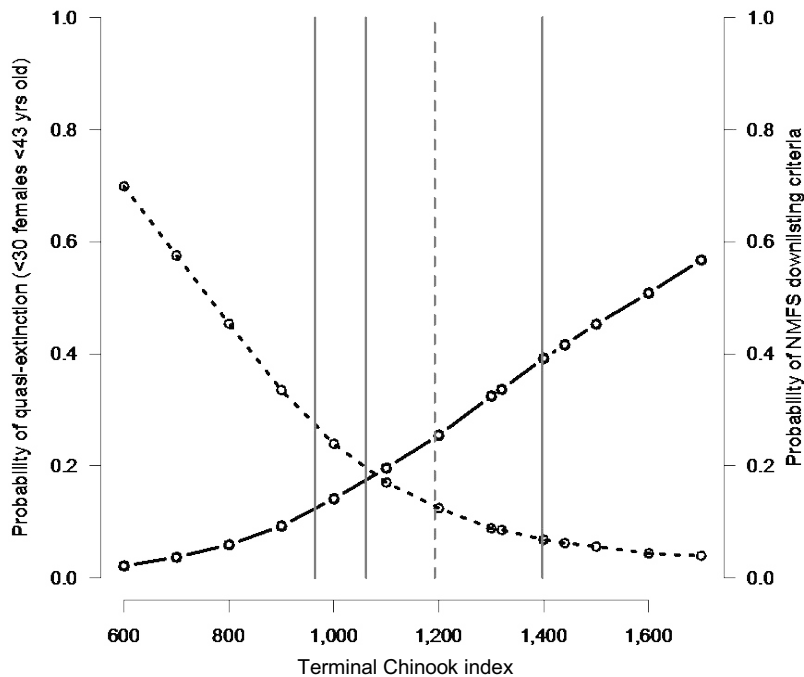
Figure 22. Illustration of how two types of recovery goals might be balanced. A measure of risk (probability of <30 or females <43 years of age, assumed proportional to population extinction) on one y-axis, and the probability of downlisting (2.3% growth for 14 years) on the other. Both are calculated from 30-year projections. Quartiles for the terminal OCE index 1979–2010 are vertical solid gray lines and the mean terminal index is the vertical dashed gray line.

43
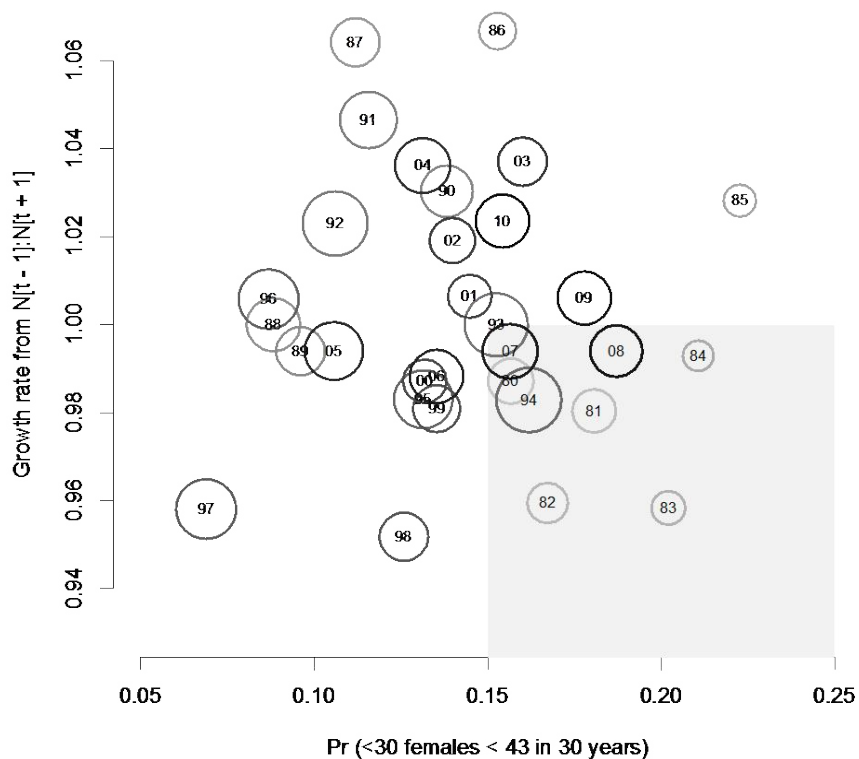
Figure 23. Illustration of how two types of recovery goals might be balanced. A measure of risk (probability of <30 females <43 years old, assumed proportional to population extinction, is calculated on the x-axis), and the realized growth rate (y-axis) indicate that the population was in its worst shape in the early 1980s. Growth rate is calculated as $1 + 0.5 \times \ln(N[t+1] / N[t-1])$. Circles are proportional to SRKW population size, with calendar years indicated.

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Appendix A: SRKW Population Summaries

This appendix contains the following 10 figures for summaries spanning 1974 to 2011:

Figure A-1. Births by pod and total births for SRKWs, 1974–2011.
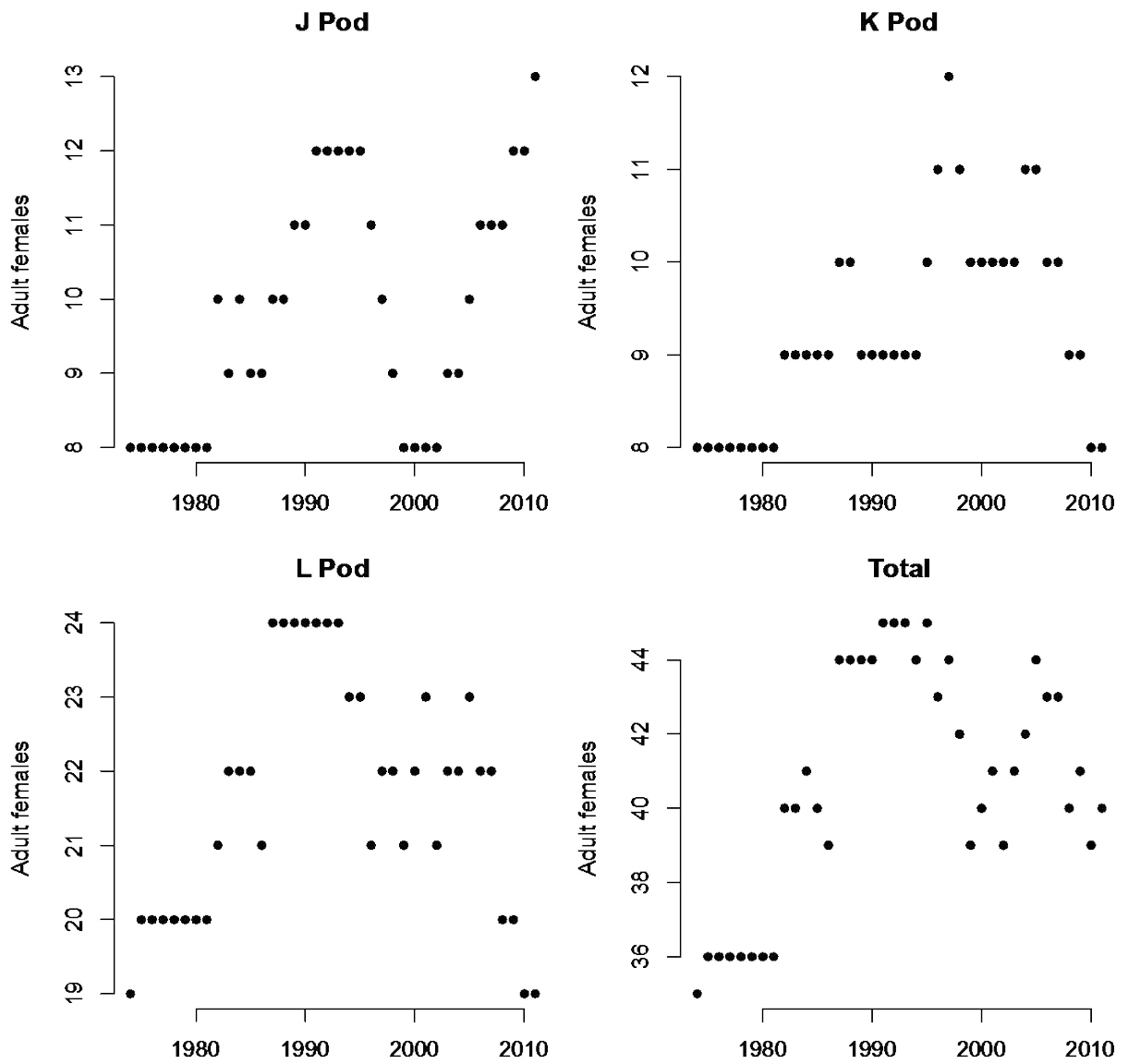
Figure A-2. Adult females (>9 years old) by pod and total adult females for SRKWs, 1974–2011.
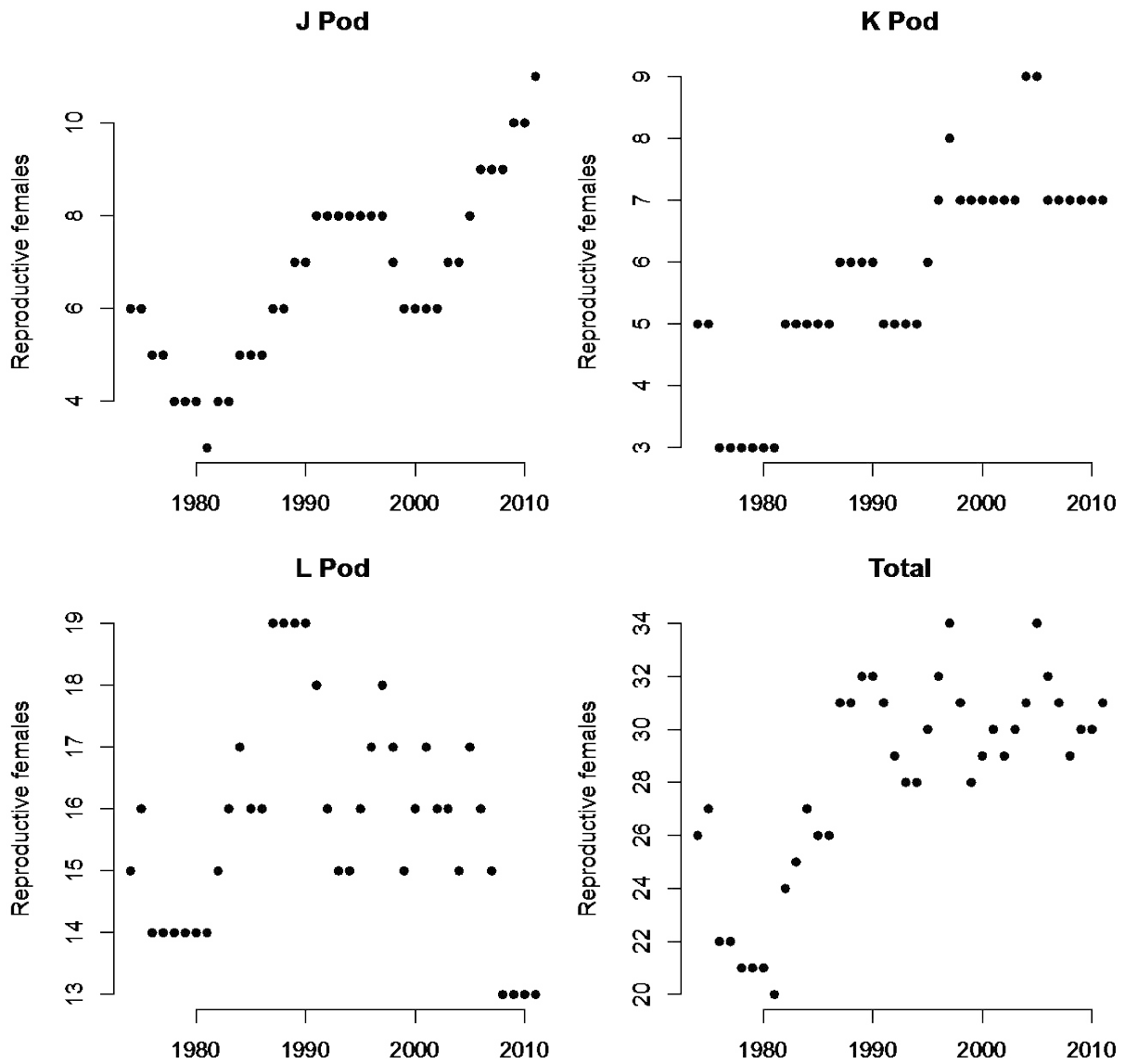
Figure A-3. Reproductive females (10–42 years old) by pod and total reproductive females for SRKWs, 1974–2011.

54
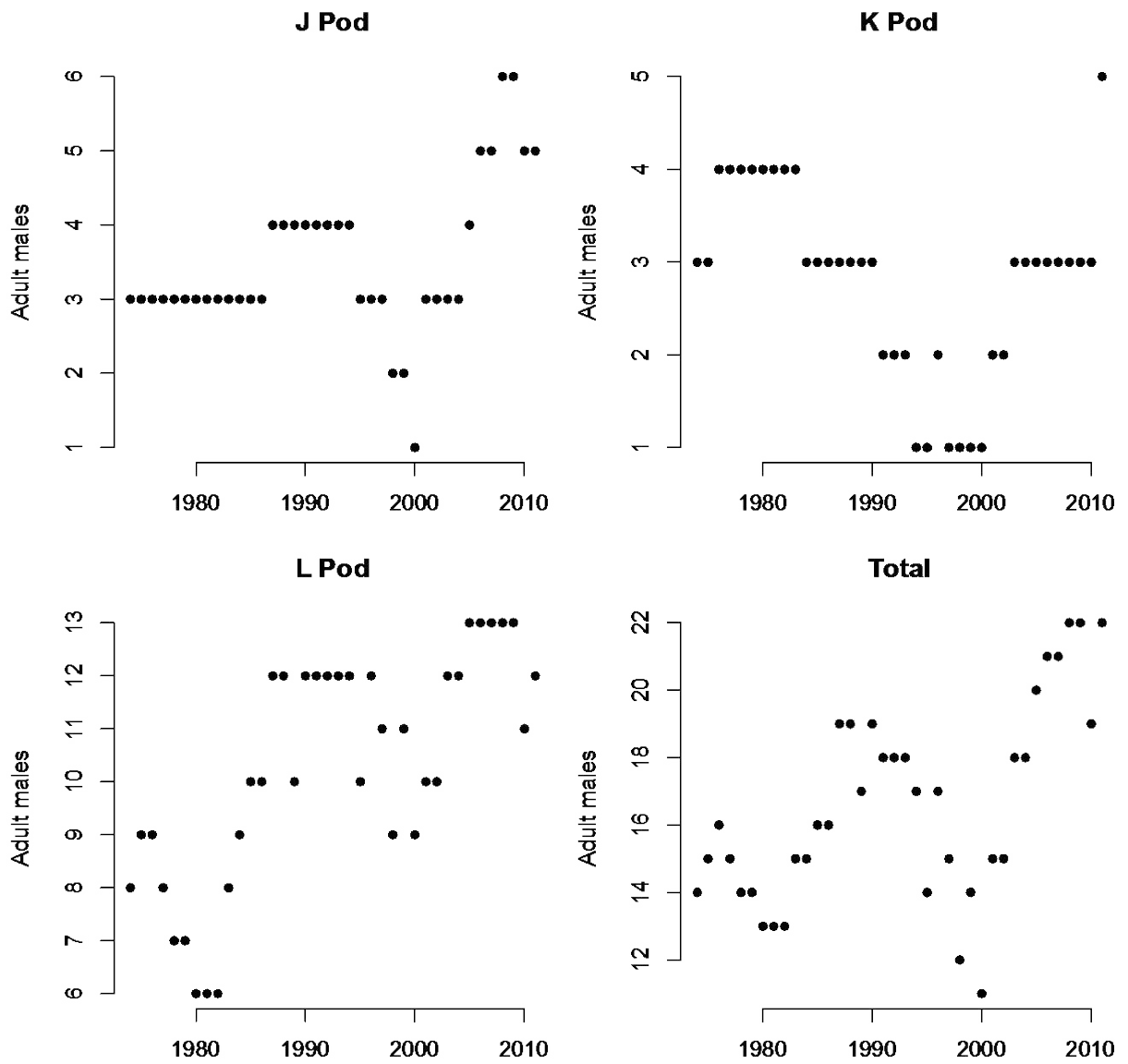
Figure A-4. Adult males (>9 years old) by pod and total adult males for SRKWs, 1974–2011.
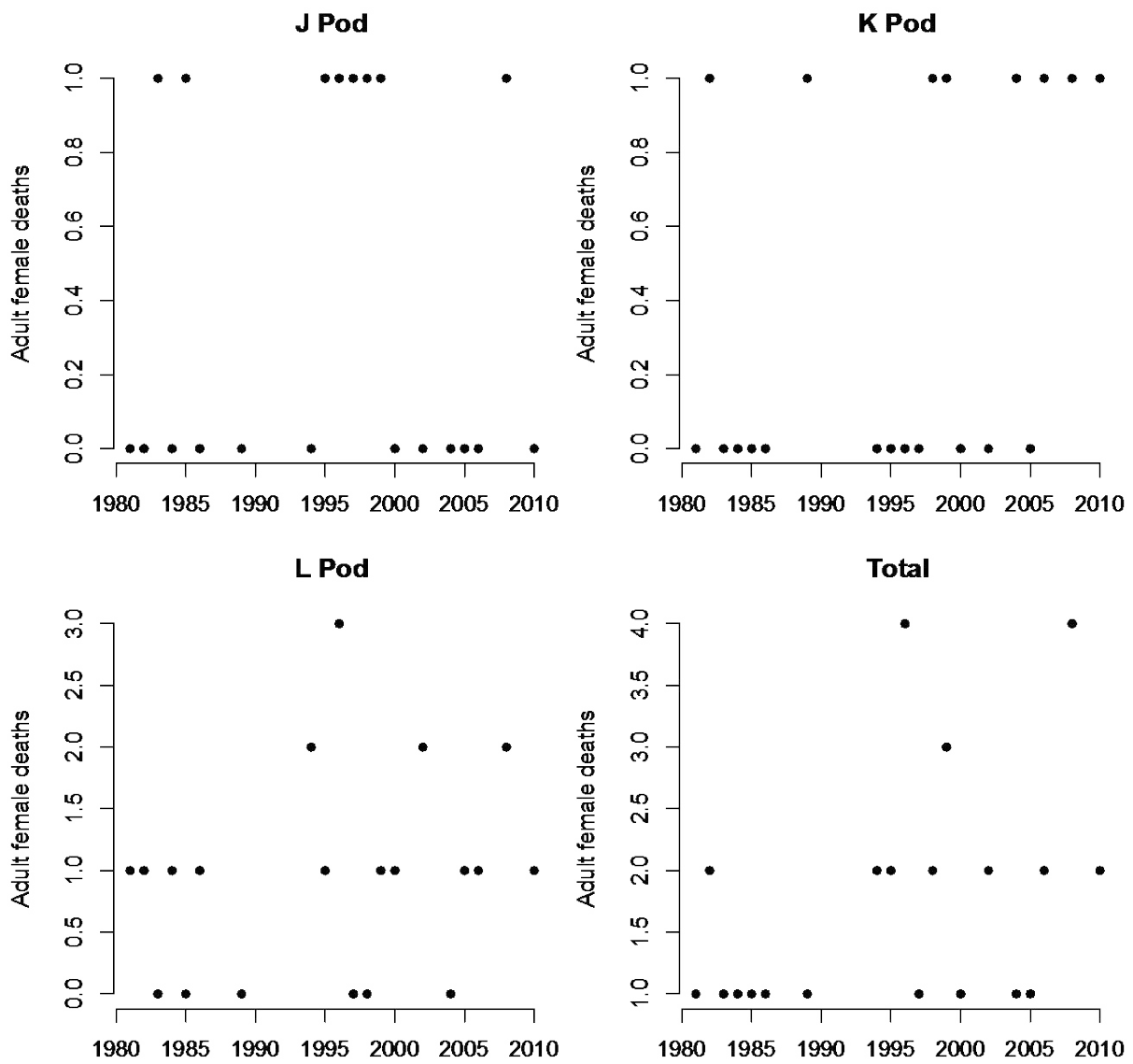
Figure A-5. Adult female (>9 years old) deaths by pod and total adult female deaths for SRKWs, 1974–2011.

AR043940
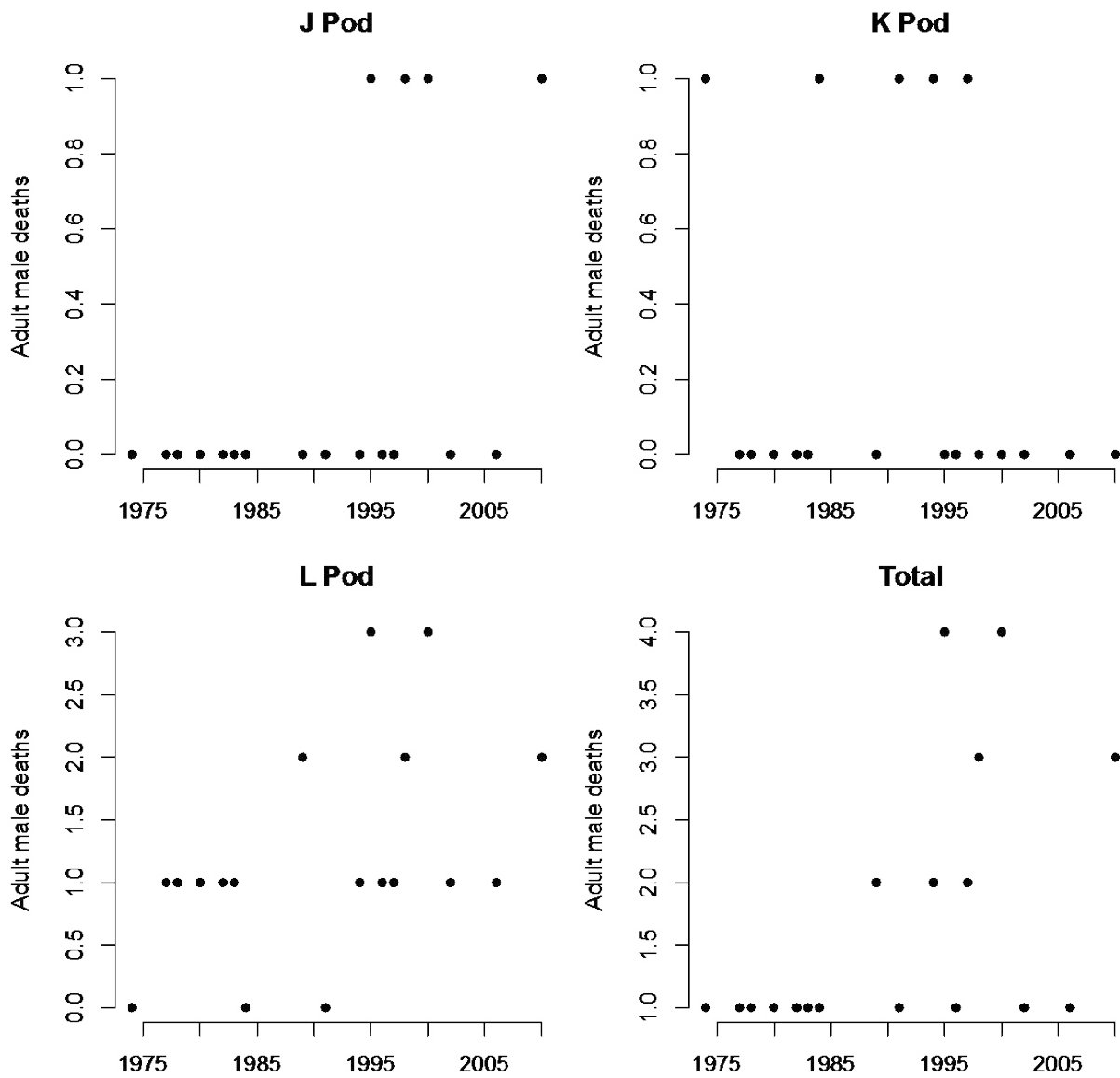
Figure A-6. Adult male (>9 years old) deaths by pod and total adult male deaths for SRKWs, 1974–2011.

AR043941
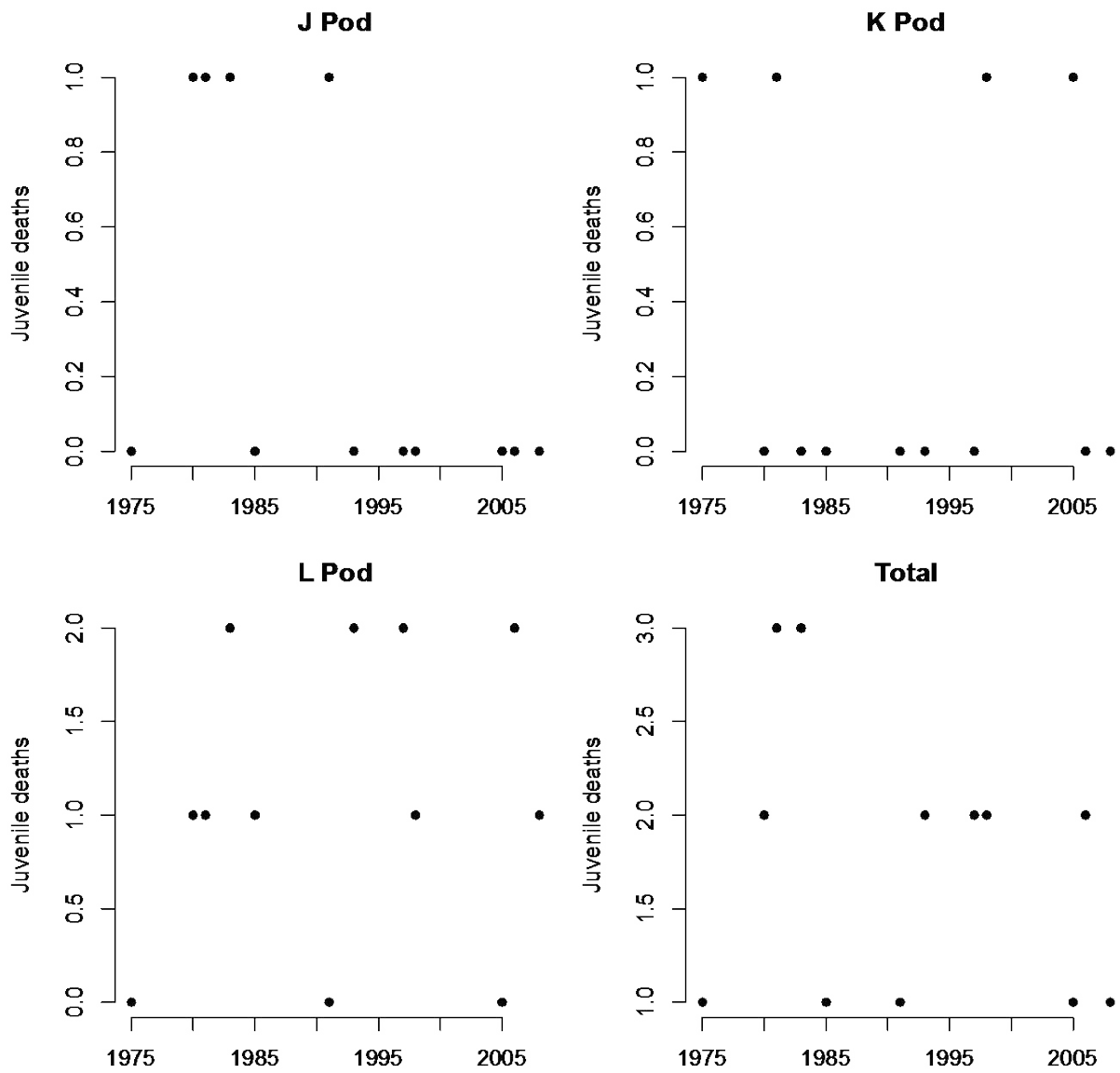
Figure A-7. Juvenile (<10 years old) deaths by pod and total juvenile deaths for SRKWs, 1974–2011.
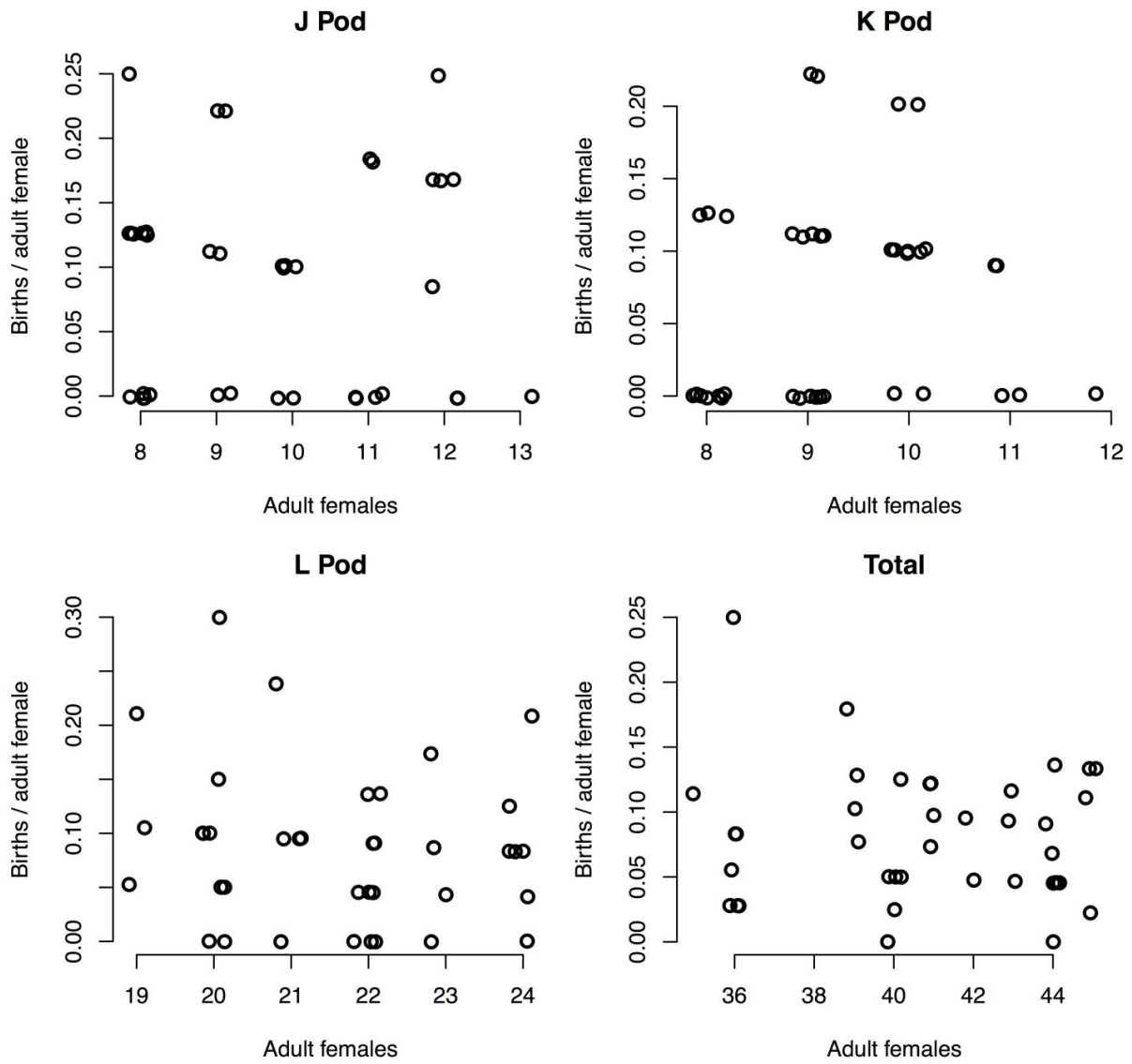
Figure A-8. Births per adult female (>9 years old) by pod and total births per adult female for SRKWs, 1974–2011.
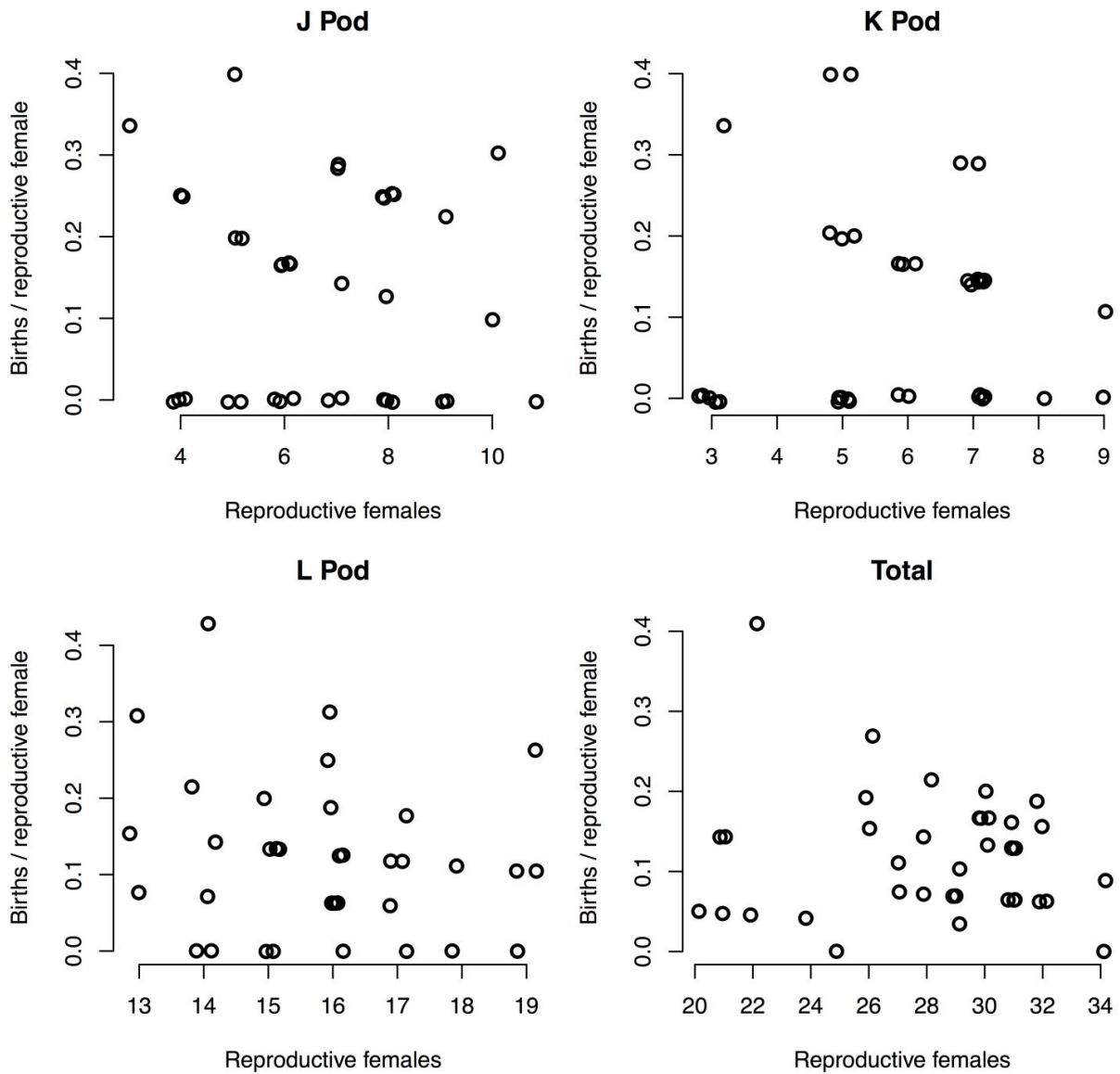
Figure A-9. Births per reproductive female (10–40 years old) by pod and total births per reproductive female for SRKWs, 1974–2011.

60
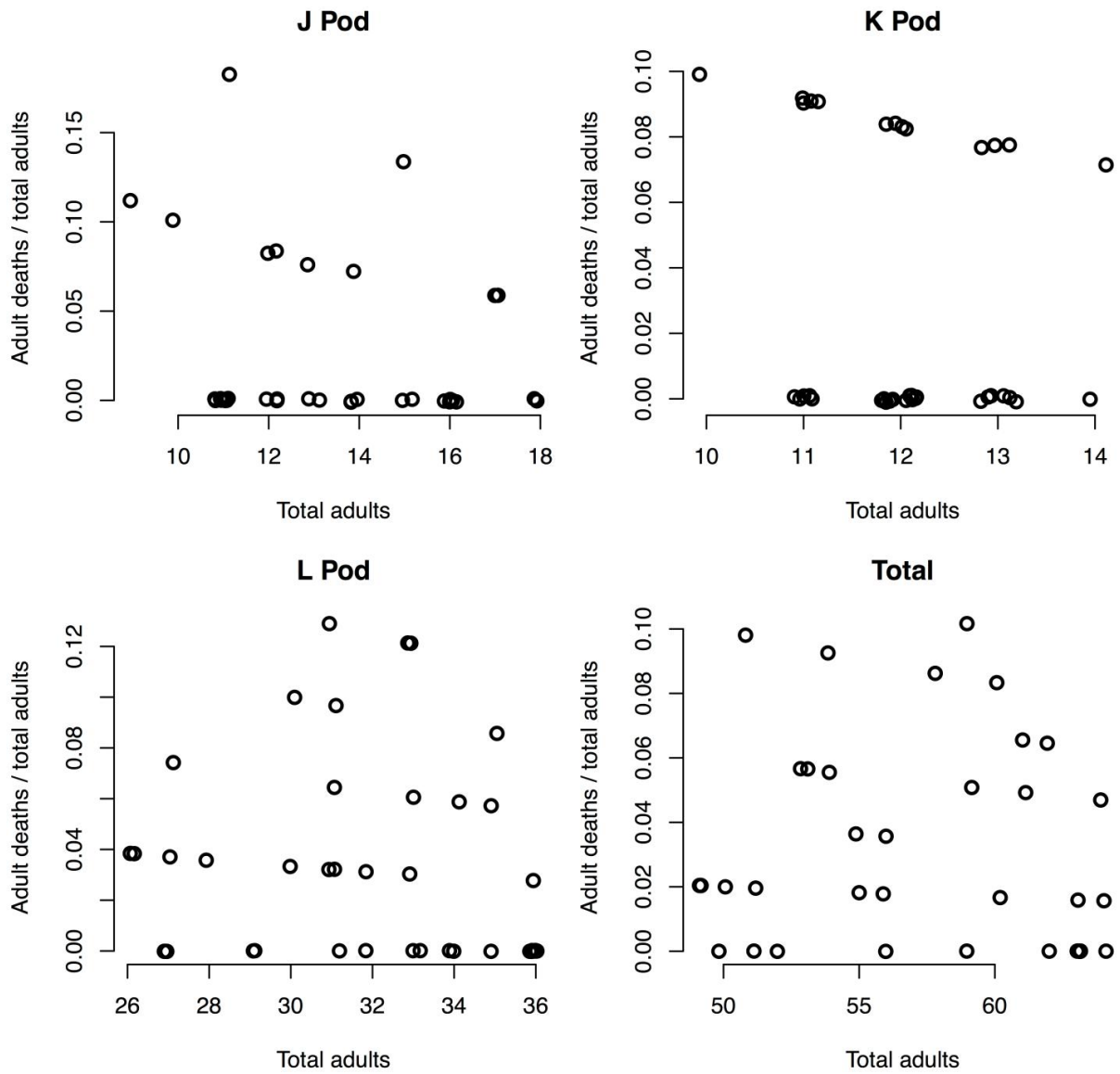
Figure A-10. Total adult (≥10 years old) death rate versus total adults by pod and total adult death rate versus total adults for SRKWs, 1974–2011.

Appendix B: Output from SRKW Population Projections

This appendix contains the following eight tables projecting over 30 years into the future across different levels of Chinook salmon abundance:

Table B-1. Mean and SD of J Pod.

Table B-2. Mean and SD of expected changes in J Pod relative to the status quo.

Table B-3. Mean and SD of K Pod.

Table B-4. Mean and SD of expected changes in K Pod relative to the status quo.

Table B-5. Mean and SD of L Pod.

Table B-6. Mean and SD of expected changes in L Pod relative to the status quo.

Table B-7. Mean and SD of the total SRKW population.

Table B-8. Mean and SD of expected changes in the total SRKW population relative to the status quo.

Table B-1. Mean and SD of J Pod over 30 years into the future across different levels of simulated fall Chinook salmon terminal run abundance (1,000s of fish).

Year	600		800		1,000		1,200		1,400		1,600	
	Mean	SD	Mean	SD	Mean	SD	Mean	SD	Mean	SD	Mean	SD
1	27.0	1.6	27.4	1.4	27.6	1.4	27.8	1.3	27.9	1.3	28.0	1.2
2	27.0	2.2	27.7	2.1	28.2	1.9	28.5	1.8	28.8	1.8	29.0	1.7
3	26.9	2.9	28.0	2.7	28.7	2.5	29.3	2.4	29.7	2.4	30.0	2.3
4	26.8	3.5	28.3	3.2	29.3	3.0	30.1	2.9	30.6	2.9	31.0	2.8
5	26.9	4.0	28.7	3.7	30.0	3.5	30.9	3.4	31.6	3.4	32.1	3.3
6	26.9	4.5	29.0	4.2	30.6	4.0	31.7	3.9	32.6	3.8	33.2	3.8
7	26.9	5.0	29.3	4.6	31.2	4.4	32.5	4.4	33.6	4.3	34.3	4.3
8	26.7	5.4	29.5	5.1	31.7	4.9	33.3	4.9	34.6	4.8	35.5	4.8
9	26.7	5.9	29.8	5.5	32.3	5.3	34.1	5.4	35.6	5.3	36.6	5.3
10	26.5	6.3	30.0	5.9	32.8	5.8	34.8	5.8	36.5	5.8	37.7	5.8
11	26.4	6.7	30.2	6.4	33.3	6.2	35.6	6.3	37.5	6.3	38.9	6.4
12	26.3	7.1	30.4	6.8	33.8	6.7	36.4	6.8	38.5	6.9	40.0	6.9
13	26.3	7.5	30.7	7.2	34.3	7.2	37.2	7.3	39.5	7.4	41.2	7.5
14	26.3	7.9	30.9	7.7	34.9	7.7	38.0	7.8	40.5	8.0	42.5	8.1
15	26.2	8.3	31.2	8.1	35.4	8.2	38.8	8.4	41.7	8.6	43.7	8.8
16	26.2	8.7	31.5	8.6	36.0	8.7	39.7	9.0	42.8	9.2	45.1	9.5
17	26.3	9.2	31.8	9.1	36.6	9.3	40.6	9.6	43.9	9.9	46.4	10.2
18	26.4	9.6	32.2	9.6	37.3	9.9	41.6	10.3	45.1	10.7	47.8	11.0
19	26.4	10.0	32.5	10.2	37.9	10.5	42.5	11.0	46.3	11.5	49.2	11.8
20	26.5	10.5	32.9	10.7	38.6	11.2	43.5	11.7	47.6	12.3	50.7	12.7
21	26.4	11.0	33.2	11.3	39.3	11.9	44.5	12.5	48.8	13.2	52.2	13.7
22	26.5	11.4	33.6	11.9	40.0	12.7	45.5	13.4	50.1	14.2	53.7	14.7
23	26.6	11.9	34.0	12.6	40.7	13.4	46.5	14.2	51.5	15.1	55.4	15.8
24	26.6	12.4	34.4	13.2	41.5	14.2	47.6	15.2	52.9	16.2	57.0	17.0
25	26.7	12.9	34.8	13.9	42.2	14.9	48.7	16.1	54.4	17.3	58.7	18.2
26	26.8	13.4	35.1	14.5	42.9	15.8	49.8	17.1	55.8	18.4	60.4	19.4
27	26.8	13.9	35.5	15.2	43.7	16.6	50.9	18.1	57.2	19.6	62.2	20.7
28	26.8	14.4	35.8	15.9	44.4	17.4	52.0	19.1	58.7	20.8	64.0	22.0
29	26.8	14.8	36.1	16.5	45.1	18.3	53.1	20.2	60.1	22.0	65.7	23.4
30	26.7	15.3	36.3	17.2	45.7	19.2	54.1	21.3	61.5	23.3	67.4	24.9

Table B-2. Mean and SD of expected changes in J Pod, relative to the status quo (1.2 million Chinook), over 30 years into the future across different levels of Chinook salmon abundance.

Year	600 Mean	600 SD	800 Mean	800 SD	1,000 Mean	1,000 SD	1,400 Mean	1,400 SD	1,600 Mean	1,600 SD
1	−0.8	1.6	−0.4	1.4	−0.2	1.4	0.1	1.3	0.2	1.2
2	−1.5	2.2	−0.8	2.1	−0.3	1.9	0.2	1.8	0.4	1.7
3	−2.4	2.9	−1.3	2.7	−0.6	2.5	0.4	2.4	0.7	2.3
4	−3.2	3.5	−1.8	3.2	−0.7	3.0	0.6	2.9	0.9	2.8
5	−4.0	4.0	−2.2	3.7	−0.9	3.5	0.7	3.4	1.2	3.3
6	−4.8	4.5	−2.7	4.2	−1.1	4.0	0.9	3.8	1.5	3.8
7	−5.6	5.0	−3.2	4.6	−1.3	4.4	1.1	4.3	1.8	4.3
8	−6.6	5.4	−3.8	5.1	−1.6	4.9	1.3	4.8	2.2	4.8
9	−7.4	5.9	−4.3	5.5	−1.8	5.3	1.5	5.3	2.5	5.3
10	−8.3	6.3	−4.9	5.9	−2.1	5.8	1.7	5.8	2.9	5.8
11	−9.2	6.7	−5.4	6.4	−2.4	6.2	1.9	6.3	3.3	6.4
12	−10.1	7.1	−6.0	6.8	−2.6	6.7	2.1	6.9	3.6	6.9
13	−10.9	7.5	−6.5	7.2	−2.9	7.2	2.3	7.4	4.0	7.5
14	−11.7	7.9	−7.1	7.7	−3.2	7.7	2.5	8.0	4.5	8.1
15	−12.6	8.3	−7.6	8.1	−3.4	8.2	2.8	8.6	4.9	8.8
16	−13.5	8.7	−8.2	8.6	−3.7	8.7	3.0	9.2	5.3	9.5
17	−14.4	9.2	−8.8	9.1	−4.0	9.3	3.3	9.9	5.8	10.2
18	−15.2	9.6	−9.4	9.6	−4.3	9.9	3.5	10.7	6.2	11.0
19	−16.1	10.0	−10.0	10.2	−4.5	10.5	3.8	11.5	6.7	11.8
20	−17.0	10.5	−10.6	10.7	−4.9	11.2	4.1	12.3	7.2	12.7
21	−18.0	11.0	−11.3	11.3	−5.2	11.9	4.3	13.2	7.7	13.7
22	−19.0	11.4	−11.9	11.9	−5.5	12.7	4.6	14.2	8.2	14.7
23	−20.0	11.9	−12.6	12.6	−5.8	13.4	5.0	15.1	8.8	15.8
24	−21.0	12.4	−13.3	13.2	−6.1	14.2	5.3	16.2	9.4	17.0
25	−22.0	12.9	−14.0	13.9	−6.5	14.9	5.7	17.3	10.0	18.2
26	−23.1	13.4	−14.7	14.5	−6.9	15.8	6.0	18.4	10.6	19.4
27	−24.1	13.9	−15.5	15.2	−7.2	16.6	6.3	19.6	11.3	20.7
28	−25.2	14.4	−16.2	15.9	−7.6	17.4	6.7	20.8	11.9	22.0
29	−26.3	14.8	−17.0	16.5	−8.0	18.3	7.1	22.0	12.6	23.4
30	−27.4	15.3	−17.8	17.2	−8.4	19.2	7.4	23.3	13.3	24.9

Table B-3. Mean and SD of K Pod over 30 years into the future across different levels of simulated Chinook salmon abundance.

Year	600 Mean	600 SD	800 Mean	800 SD	1,000 Mean	1,000 SD	1,200 Mean	1,200 SD	1,400 Mean	1,400 SD	1,600 Mean	1,600 SD
1	18.0	1.4	18.3	1.3	18.5	1.2	18.6	1.1	18.7	1.1	18.8	1.1
2	17.9	1.8	18.4	1.7	18.8	1.6	19.0	1.5	19.2	1.4	19.4	1.4
3	17.7	2.3	18.4	2.1	19.0	2.0	19.4	1.9	19.7	1.8	19.9	1.8
4	17.4	2.6	18.4	2.4	19.2	2.3	19.7	2.2	20.1	2.1	20.4	2.1
5	17.1	2.9	18.3	2.7	19.2	2.6	19.9	2.5	20.4	2.4	20.8	2.4
6	16.6	3.2	18.0	2.9	19.1	2.8	20.0	2.7	20.6	2.7	21.1	2.6
7	16.2	3.4	17.8	3.2	19.1	3.1	20.1	3.0	20.9	2.9	21.4	2.9
8	15.9	3.7	17.6	3.4	19.1	3.3	20.3	3.2	21.1	3.1	21.8	3.1
9	15.6	3.9	17.5	3.6	19.1	3.5	20.4	3.4	21.4	3.4	22.1	3.4
10	15.3	4.0	17.4	3.8	19.1	3.7	20.5	3.7	21.6	3.6	22.5	3.6
11	15.0	4.2	17.2	4.0	19.1	3.9	20.7	3.9	21.9	3.8	22.8	3.8
12	14.7	4.4	17.1	4.2	19.1	4.1	20.8	4.1	22.2	4.1	23.2	4.1
13	14.4	4.6	16.9	4.4	19.1	4.4	21.0	4.4	22.4	4.3	23.5	4.3
14	14.1	4.7	16.7	4.6	19.1	4.6	21.1	4.6	22.7	4.6	23.9	4.6
15	13.7	4.9	16.4	4.8	19.0	4.9	21.2	4.9	22.9	5.0	24.2	5.0
16	13.4	5.1	16.3	5.1	19.0	5.1	21.3	5.2	23.2	5.3	24.6	5.3
17	13.1	5.3	16.1	5.3	19.0	5.4	21.5	5.5	23.4	5.7	25.1	5.7
18	12.9	5.5	16.0	5.6	19.0	5.7	21.6	5.9	23.7	6.0	25.4	6.1
19	12.7	5.6	15.8	5.8	18.9	6.0	21.7	6.2	24.0	6.4	25.8	6.5
20	12.4	5.8	15.6	6.0	18.8	6.2	21.7	6.5	24.2	6.8	26.1	6.9
21	12.1	5.9	15.5	6.2	18.8	6.5	21.8	6.8	24.4	7.2	26.5	7.4
22	11.9	6.1	15.3	6.5	18.8	6.8	22.0	7.2	24.6	7.6	26.9	7.9
23	11.7	6.3	15.2	6.7	18.8	7.2	22.1	7.6	24.9	8.0	27.3	8.4
24	11.6	6.4	15.1	7.0	18.8	7.5	22.2	8.0	25.2	8.5	27.7	8.9
25	11.4	6.6	15.0	7.2	18.9	7.8	22.4	8.4	25.5	8.9	28.2	9.4
26	11.3	6.8	15.0	7.4	18.9	8.2	22.6	8.8	25.9	9.4	28.7	10.0
27	11.2	7.0	14.9	7.7	19.0	8.5	22.8	9.3	26.2	9.9	29.2	10.5
28	11.0	7.1	14.9	8.0	19.0	8.9	23.0	9.7	26.5	10.5	29.7	11.1
29	10.9	7.3	14.8	8.2	19.1	9.2	23.3	10.2	26.9	11.0	30.2	11.7
30	10.9	7.5	14.8	8.5	19.2	9.6	23.5	10.7	27.3	11.6	30.7	12.4

Table B-4. Mean and SD of expected changes in K Pod, relative to the status quo (1.2 million Chinook), over 30 years into the future across different levels of Chinook salmon abundance.

Year	600		800		1,000		1,400		1,600	
	Mean	SD	Mean	SD	Mean	SD	Mean	SD	Mean	SD
1	−0.6	1.4	−0.3	1.3	−0.1	1.2	0.1	1.1	0.2	1.1
2	−1.1	1.8	−0.6	1.7	−0.3	1.6	0.2	1.4	0.3	1.4
3	−1.7	2.3	−1.0	2.1	−0.4	2.0	0.3	1.8	0.5	1.8
4	−2.3	2.6	−1.3	2.4	−0.5	2.3	0.4	2.1	0.7	2.1
5	−2.8	2.9	−1.6	2.7	−0.7	2.6	0.5	2.4	0.9	2.4
6	−3.4	3.2	−2.0	2.9	−0.9	2.8	0.6	2.7	1.1	2.6
7	−3.9	3.4	−2.3	3.2	−1.0	3.1	0.8	2.9	1.3	2.9
8	−4.4	3.7	−2.6	3.4	−1.2	3.3	0.9	3.1	1.5	3.1
9	−4.9	3.9	−2.9	3.6	−1.3	3.5	1.0	3.4	1.7	3.4
10	−5.3	4.0	−3.2	3.8	−1.4	3.7	1.1	3.6	1.9	3.6
11	−5.7	4.2	−3.5	4.0	−1.6	3.9	1.2	3.8	2.1	3.8
12	−6.1	4.4	−3.7	4.2	−1.7	4.1	1.3	4.1	2.4	4.1
13	−6.6	4.6	−4.1	4.4	−1.8	4.4	1.5	4.3	2.6	4.3
14	−7.0	4.7	−4.4	4.6	−2.0	4.6	1.6	4.6	2.8	4.6
15	−7.5	4.9	−4.7	4.8	−2.2	4.9	1.7	5.0	3.1	5.0
16	−7.9	5.1	−5.0	5.1	−2.3	5.1	1.9	5.3	3.3	5.3
17	−8.3	5.3	−5.3	5.3	−2.5	5.4	2.0	5.7	3.6	5.7
18	−8.7	5.5	−5.6	5.6	−2.6	5.7	2.1	6.0	3.9	6.1
19	−9.0	5.6	−5.8	5.8	−2.8	6.0	2.3	6.4	4.1	6.5
20	−9.3	5.8	−6.1	6.0	−2.9	6.2	2.4	6.8	4.4	6.9
21	−9.7	5.9	−6.4	6.2	−3.0	6.5	2.6	7.2	4.6	7.4
22	−10.0	6.1	−6.6	6.5	−3.2	6.8	2.7	7.6	4.9	7.9
23	−10.3	6.3	−6.9	6.7	−3.3	7.2	2.9	8.0	5.2	8.4
24	−10.6	6.4	−7.1	7.0	−3.4	7.5	3.0	8.5	5.5	8.9
25	−11.0	6.6	−7.4	7.2	−3.5	7.8	3.1	8.9	5.8	9.4
26	−11.3	6.8	−7.6	7.4	−3.7	8.2	3.3	9.4	6.1	10.0
27	−11.6	7.0	−7.9	7.7	−3.8	8.5	3.4	9.9	6.4	10.5
28	−12.0	7.1	−8.2	8.0	−4.0	8.9	3.5	10.5	6.6	11.1
29	−12.3	7.3	−8.4	8.2	−4.1	9.2	3.7	11.0	6.9	11.7
30	−12.6	7.5	−8.7	8.5	−4.3	9.6	3.8	11.6	7.2	12.4

67

Table B-5. Mean and SD of L Pod over 30 years into the future across different levels of simulated Chinook salmon abundance.

Year	600 Mean	600 SD	800 Mean	800 SD	1,000 Mean	1,000 SD	1,200 Mean	1,200 SD	1,400 Mean	1,400 SD	1,600 Mean	1,600 SD
1	38.9	2.4	39.9	2.2	40.6	1.9	41.1	1.8	41.5	1.7	41.8	1.6
2	37.1	3.5	38.9	3.1	40.3	2.8	41.4	2.6	42.1	2.4	42.7	2.3
3	35.5	4.3	38.0	3.8	40.0	3.5	41.5	3.3	42.6	3.2	43.5	3.0
4	33.9	4.9	37.0	4.4	39.6	4.1	41.5	3.9	43.0	3.7	44.1	3.6
5	32.2	5.5	35.9	4.9	39.0	4.6	41.3	4.4	43.2	4.3	44.7	4.2
6	30.7	5.9	34.9	5.3	38.3	5.0	41.1	4.9	43.4	4.8	45.1	4.7
7	29.3	6.2	33.8	5.6	37.7	5.4	40.8	5.2	43.4	5.2	45.4	5.2
8	27.8	6.5	32.6	6.0	36.9	5.7	40.4	5.6	43.3	5.6	45.6	5.6
9	26.6	6.7	31.6	6.2	36.2	6.0	40.0	6.0	43.2	6.0	45.8	6.1
10	25.4	6.9	30.6	6.4	35.4	6.2	39.5	6.2	43.0	6.3	45.8	6.4
11	24.2	7.0	29.6	6.6	34.6	6.5	39.0	6.5	42.8	6.7	45.9	6.8
12	23.1	7.2	28.7	6.8	34.0	6.7	38.7	6.8	42.7	7.0	45.9	7.2
13	22.2	7.4	27.9	7.1	33.4	6.9	38.3	7.2	42.6	7.4	46.1	7.7
14	21.4	7.5	27.2	7.3	32.9	7.3	38.1	7.5	42.6	7.8	46.4	8.2
15	20.8	7.7	26.7	7.6	32.6	7.6	38.0	7.9	42.8	8.3	46.8	8.7
16	20.2	7.9	26.3	7.8	32.3	8.0	38.0	8.4	43.1	8.9	47.3	9.3
17	19.7	8.2	25.8	8.2	32.2	8.4	38.0	8.9	43.4	9.5	47.8	10.0
18	19.2	8.4	25.5	8.5	32.0	8.8	38.1	9.4	43.8	10.1	48.5	10.8
19	18.7	8.6	25.1	8.8	31.8	9.2	38.2	10.0	44.1	10.8	49.0	11.6
20	18.2	8.8	24.7	9.1	31.6	9.6	38.3	10.5	44.4	11.4	49.6	12.4
21	17.8	9.0	24.3	9.5	31.5	10.1	38.4	11.1	44.8	12.2	50.3	13.2
22	17.3	9.2	24.0	9.8	31.4	10.6	38.4	11.7	45.1	12.9	51.0	14.1
23	16.9	9.4	23.6	10.1	31.1	11.0	38.5	12.3	45.4	13.6	51.5	14.9
24	16.4	9.5	23.2	10.4	30.9	11.4	38.4	12.9	45.7	14.3	52.0	15.7
25	15.8	9.6	22.8	10.6	30.6	11.8	38.3	13.4	45.9	15.0	52.5	16.6
26	15.4	9.7	22.4	10.9	30.3	12.2	38.3	14.0	46.1	15.8	53.0	17.5
27	14.9	9.8	22.0	11.1	30.0	12.6	38.2	14.5	46.4	16.5	53.5	18.4
28	14.5	9.9	21.6	11.4	29.7	13.0	38.2	15.1	46.6	17.3	54.0	19.4
29	14.0	10.0	21.1	11.6	29.4	13.4	38.1	15.7	46.8	18.1	54.5	20.3
30	13.6	10.0	20.8	11.8	29.2	13.8	38.1	16.3	47.1	18.9	55.0	21.4

68

Table B-6. Mean and SD of expected changes in L Pod, relative to the status quo (1.2 million Chinook), over 30 years into the future across different levels of Chinook salmon abundance.

	600		800		1,000		1,400		1,600	
Year	Mean	SD	Mean	SD	Mean	SD	Mean	SD	Mean	SD
1	−2.3	2.4	−1.3	2.2	−0.5	1.9	0.4	1.7	0.7	1.6
2	−4.3	3.5	−2.4	3.1	−1.1	2.8	0.8	2.4	1.3	2.3
3	−6.0	4.3	−3.5	3.8	−1.5	3.5	1.1	3.2	2.0	3.0
4	−7.6	4.9	−4.5	4.4	−1.9	4.1	1.5	3.7	2.6	3.6
5	−9.1	5.5	−5.5	4.9	−2.4	4.6	1.9	4.3	3.3	4.2
6	−10.4	5.9	−6.3	5.3	−2.8	5.0	2.2	4.8	4.0	4.7
7	−11.6	6.2	−7.0	5.6	−3.1	5.4	2.6	5.2	4.6	5.2
8	−12.6	6.5	−7.8	6.0	−3.5	5.7	2.9	5.6	5.2	5.6
9	−13.4	6.7	−8.4	6.2	−3.9	6.0	3.2	6.0	5.7	6.1
10	−14.1	6.9	−8.9	6.4	−4.1	6.2	3.5	6.3	6.3	6.4
11	−14.9	7.0	−9.5	6.6	−4.4	6.5	3.8	6.7	6.8	6.8
12	−15.5	7.2	−10.0	6.8	−4.7	6.7	4.0	7.0	7.3	7.2
13	−16.1	7.4	−10.4	7.1	−4.9	6.9	4.3	7.4	7.8	7.7
14	−16.7	7.5	−10.9	7.3	−5.2	7.3	4.5	7.8	8.3	8.2
15	−17.2	7.7	−11.3	7.6	−5.4	7.6	4.8	8.3	8.8	8.7
16	−17.8	7.9	−11.8	7.8	−5.7	8.0	5.0	8.9	9.3	9.3
17	−18.3	8.2	−12.2	8.2	−5.9	8.4	5.4	9.5	9.8	10.0
18	−18.9	8.4	−12.6	8.5	−6.1	8.8	5.6	10.1	10.3	10.8
19	−19.5	8.6	−13.1	8.8	−6.4	9.2	5.9	10.8	10.8	11.6
20	−20.0	8.8	−13.6	9.1	−6.6	9.6	6.2	11.4	11.4	12.4
21	−20.6	9.0	−14.0	9.5	−6.8	10.1	6.4	12.2	11.9	13.2
22	−21.1	9.2	−14.4	9.8	−7.1	10.6	6.7	12.9	12.5	14.1
23	−21.6	9.4	−14.8	10.1	−7.3	11.0	7.0	13.6	13.0	14.9
24	−22.0	9.5	−15.2	10.4	−7.5	11.4	7.3	14.3	13.6	15.7
25	−22.5	9.6	−15.6	10.6	−7.8	11.8	7.6	15.0	14.2	16.6
26	−22.9	9.7	−15.9	10.9	−8.0	12.2	7.8	15.8	14.7	17.5
27	−23.3	9.8	−16.3	11.1	−8.3	12.6	8.1	16.5	15.3	18.4
28	−23.7	9.9	−16.6	11.4	−8.5	13.0	8.4	17.3	15.8	19.4
29	−24.1	10.0	−17.0	11.6	−8.7	13.4	8.7	18.1	16.4	20.3
30	−24.5	10.0	−17.3	11.8	−8.9	13.8	9.0	18.9	16.9	21.4

69

Table B-7. Mean and SD of the total SRKW population over 30 years into the future across different levels of simulated Chinook salmon abundance.

Year	600 Mean	600 SD	800 Mean	800 SD	1,000 Mean	1,000 SD	1,200 Mean	1,200 SD	1,400 Mean	1,400 SD	1,600 Mean	1,600 SD
1	83.9	3.7	85.6	3.2	86.7	3.0	87.6	2.8	88.2	2.7	88.6	2.6
2	82.0	5.4	85.0	4.8	87.3	4.4	88.9	4.2	90.1	4.0	91.0	3.9
3	80.0	7.2	84.4	6.3	87.7	5.8	90.2	5.6	92.0	5.4	93.4	5.3
4	78.2	8.5	83.7	7.5	88.1	7.0	91.3	6.8	93.7	6.7	95.6	6.6
5	76.1	9.6	82.8	8.5	88.1	8.1	92.1	7.9	95.2	7.8	97.5	7.8
6	74.2	10.7	81.9	9.5	88.1	9.0	92.9	8.9	96.6	8.9	99.4	9.0
7	72.3	11.6	80.9	10.4	87.9	9.9	93.4	9.9	97.8	10.0	101.1	10.1
8	70.4	12.5	79.8	11.2	87.6	10.8	93.9	10.9	99.0	11.0	102.8	11.2
9	68.8	13.3	78.9	12.1	87.5	11.7	94.5	11.8	100.2	12.1	104.5	12.3
10	67.3	13.9	77.9	12.8	87.2	12.4	94.9	12.7	101.2	13.1	106.0	13.4
11	65.6	14.7	77.0	13.5	87.0	13.2	95.4	13.5	102.2	14.0	107.6	14.4
12	64.2	15.4	76.2	14.3	86.9	14.0	95.9	14.5	103.3	15.1	109.1	15.6
13	62.9	16.1	75.5	15.1	86.8	14.9	96.5	15.5	104.5	16.2	110.9	16.8
14	61.8	16.8	74.9	15.9	86.9	15.9	97.2	16.6	105.9	17.4	112.7	18.1
15	60.7	17.5	74.4	16.7	87.0	16.9	98.0	17.7	107.4	18.7	114.8	19.6
16	59.8	18.3	74.0	17.7	87.3	18.0	99.0	19.0	109.0	20.1	117.0	21.2
17	59.1	19.1	73.8	18.7	87.8	19.2	100.1	20.4	110.8	21.7	119.3	22.9
18	58.5	19.9	73.7	19.7	88.3	20.4	101.3	21.8	112.6	23.3	121.7	24.7
19	57.8	20.6	73.4	20.7	88.7	21.6	102.4	23.3	114.3	25.1	124.0	26.7
20	57.1	21.4	73.2	21.7	89.1	22.9	103.5	24.8	116.1	26.9	126.4	28.6
21	56.3	22.1	73.0	22.7	89.6	24.3	104.7	26.4	118.0	28.7	128.9	30.8
22	55.8	22.9	72.9	23.8	90.1	25.6	105.9	28.1	119.9	30.7	131.5	33.1
23	55.2	23.7	72.8	24.9	90.7	27.0	107.1	29.8	121.9	32.7	134.1	35.3
24	54.5	24.4	72.7	26.0	91.2	28.4	108.2	31.6	123.8	34.8	136.7	37.7
25	54.0	25.1	72.6	27.0	91.6	29.8	109.4	33.3	125.8	37.0	139.4	40.2
26	53.4	25.8	72.5	28.1	92.1	31.2	110.7	35.2	127.8	39.2	142.1	42.7
27	52.9	26.5	72.4	29.1	92.6	32.6	112.0	37.0	129.8	41.4	144.9	45.3
28	52.3	27.2	72.2	30.2	93.2	34.1	113.2	38.9	131.9	43.8	147.6	48.0
29	51.7	27.8	72.0	31.2	93.6	35.6	114.4	40.8	133.8	46.2	150.3	50.8
30	51.2	28.5	71.9	32.3	94.1	37.1	115.7	42.9	135.9	48.7	153.1	53.8

70

Table B-8. Mean and SD of expected changes in the total SRKW population, relative to the status quo (1.2 million Chinook), over 30 years into the future across different levels of Chinook salmon abundance.

Year	600 Mean	600 SD	800 Mean	800 SD	1,000 Mean	1,000 SD	1,400 Mean	1,400 SD	1,600 Mean	1,600 SD
1	−3.6	3.7	−2.0	3.2	−0.8	3.0	0.6	2.7	1.0	2.6
2	−6.9	5.4	−3.9	4.8	−1.7	4.4	1.2	4.0	2.1	3.9
3	−10.2	7.2	−5.8	6.3	−2.5	5.8	1.8	5.4	3.2	5.3
4	−13.1	8.5	−7.6	7.5	−3.2	7.0	2.4	6.7	4.3	6.6
5	−16.0	9.6	−9.3	8.5	−4.0	8.1	3.1	7.8	5.4	7.8
6	−18.6	10.7	−11.0	9.5	−4.8	9.0	3.7	8.9	6.5	9.0
7	−21.1	11.6	−12.6	10.4	−5.5	9.9	4.4	10.0	7.7	10.1
8	−23.5	12.5	−14.1	11.2	−6.3	10.8	5.0	11.0	8.9	11.2
9	−25.7	13.3	−15.6	12.1	−7.0	11.7	5.7	12.1	10.0	12.3
10	−27.7	13.9	−17.0	12.8	−7.7	12.4	6.3	13.1	11.1	13.4
11	−29.8	14.7	−18.4	13.5	−8.3	13.2	6.9	14.0	12.2	14.4
12	−31.7	15.4	−19.7	14.3	−9.0	14.0	7.4	15.1	13.2	15.6
13	−33.6	16.1	−21.0	15.1	−9.7	14.9	8.1	16.2	14.4	16.8
14	−35.5	16.8	−22.4	15.9	−10.3	15.9	8.6	17.4	15.5	18.1
15	−37.3	17.5	−23.7	16.7	−11.0	16.9	9.3	18.7	16.7	19.6
16	−39.2	18.3	−25.0	17.7	−11.7	18.0	10.0	20.1	17.9	21.2
17	−41.0	19.1	−26.3	18.7	−12.4	19.2	10.6	21.7	19.2	22.9
18	−42.8	19.9	−27.6	19.7	−13.0	20.4	11.3	23.3	20.4	24.7
19	−44.6	20.6	−28.9	20.7	−13.7	21.6	12.0	25.1	21.7	26.7
20	−46.4	21.4	−30.3	21.7	−14.4	22.9	12.6	26.9	22.9	28.6
21	−48.3	22.1	−31.7	22.7	−15.1	24.3	13.3	28.7	24.3	30.8
22	−50.1	22.9	−33.0	23.8	−15.8	25.6	14.0	30.7	25.7	33.1
23	−51.9	23.7	−34.3	24.9	−16.4	27.0	14.8	32.7	27.1	35.3
24	−53.7	24.4	−35.5	26.0	−17.1	28.4	15.6	34.8	28.5	37.7
25	−55.5	25.1	−36.9	27.0	−17.8	29.8	16.4	37.0	30.0	40.2
26	−57.3	25.8	−38.2	28.1	−18.6	31.2	17.1	39.2	31.4	42.7
27	−59.1	26.5	−39.6	29.1	−19.3	32.6	17.8	41.4	32.9	45.3
28	−60.9	27.2	−41.0	30.2	−20.1	34.1	18.6	43.8	34.4	48.0
29	−62.7	27.8	−42.4	31.2	−20.8	35.6	19.4	46.2	35.9	50.8
30	−64.5	28.5	−43.7	32.3	−21.5	37.1	20.2	48.7	37.5	53.8

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