Agenda Item E.4.a
Supplemental SRKW Workgroup Report 2 (**Electronic Only**)
November 2019

Pacific Fishery Management Council
Salmon Fishery Management Plan Impacts to Southern Resident Killer Whales

Draft Risk Assessment

November 2019

Pacific Fishery Management Council

Ad-Hoc Southern Resident Killer Whale Workgroup

Phil Anderson, Pacific Fishery Management Council
Jeromy Jording, National Marine Fisheries Service
Susan Bishop, National Marine Fisheries Service
Teresa Mongillo-Lawson, National Marine Fisheries Service
Will Satterthwaite, National Marine Fisheries Service
Eric Ward, National Marine Fisheries Service
Scott McGrew, U.S. Coast Guard
Mike Matylewich, Columbia River Intertribal Fish Commission
Nate Tyler, Makah Tribe
Tyler Gross, Quileute Tribe
Tyler Jurasin, Quinault Indian Nation
Melvinjohn Ashue, Hoh Tribe
Kyle Adicks, Washington Department of Fish & Wildlife
Derek Dapp, Washington Department of Fish & Wildlife
Chris Kern, Oregon Department of Fish & Wildlife
Craig Foster, Oregon Department of Fish & Wildlife
Lance Hebdon, Idaho Department of Fish and Game
Chris Kozfkay, Idaho Department of Fish and Game
Brett Kormos, California Department of Fish and Wildlife
Erica Meyers, California Department of Fish and Wildlife

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EXECUTIVE SUMMARY

[Pending]

1 INTRODUCTION

This report is a product of the Pacific Fishery Management Council's (PFMC or Council) ad-hoc Southern Resident Killer Whale (SRKW) Workgroup which was tasked with reassessing the effects of Council-area ocean salmon fisheries on SRKW. We first provide a brief overview of the background context, workshop process, and the role of the SRKW Workgroup. Then we assess the current status of the SRKWs, followed by describing the interactions known to occur between SRKW and salmon fisheries, leading to a general description of the Pacific Coast Salmon Fishery Management Plan (FMP). Lastly, we attempt to assess how reductions in prey through implementing the FMP may affect SRKW demographics.

1.1 Background

SRKW are listed as endangered under the Endangered Species Act (ESA) (70 FR 69903). Multiple actions along the west coast are active in conserving and recovering SRKW, particularly to address three threats to the whales that were identified in the SRKW Recovery Plan (NMFS 2008): prey limitation, vessel traffic and noise, and chemical contaminants. Fisheries affect the whales primarily through removing prey. The Council uses provisions of the FMP to make recommendations to NMFS for implementing salmon fisheries in Federal waters (3-200 nautical miles) off the coast of Washington, Oregon, and California. The effects on SRKW of implementing the FMP, *i.e.*, prey removal and the potential for interaction between fishing gear and vessels, were last consulted under the ESA per Section 7(a)(2) by NMFS in 2009 (NMFS 2009). That consultation described the effects on the amount of prey available to SRKW and the potential for interactions between fishing gear and vessels. In that opinion, NMFS concluded Council fisheries did not jeopardize the survival and recovery of SRKW.

Since the 2009 consultation was completed, new information is available on SRKW and their relationship to salmon prey species, and in March of 2019, NMFS announced plans to reinitiate consultation on the implementation of the FMP which it did on April 12, 2019. Subsequently, at its April 2019 meeting, the Council formed the ad-hoc SRKW workgroup (Workgroup) to reassess the effects of Council-area ocean salmon fisheries on the Chinook salmon prey base of SRKW, and depending on the results, develop a long-term approach that may include proposed conservation measure(s) or management tool(s) that limit PFMC fishery impacts to prey availability for SRKW relative to implementing the FMP.

The Workgroup met numerous times during the course of 2019 in order to develop the risk assessment approach contained in this report, and all meetings were open to the public. A detailed list of Workgroup meetings and presentations can be found online at: https://www.fisheries.noaa.gov/west-coast/southern-resident-killer-whales-and-fisheries-interaction-workgroup

1.2 Purpose and Need

Chinook salmon, the whales' primary prey, are important to SRKW survival and recovery. Any activities that affect the abundance of Chinook salmon available to SRKW have the potential to impact the survival and population growth of the whales. Fisheries can reduce the prey available to the whales and in some cases can interfere directly with their feeding. Insufficient prey can impact their energetics (causing them to search more for fewer prey), health (decreasing their body condition), and reproduction (reducing fecundity and calf survival).

NMFS consulted on the effects of Council fisheries under the ESA in 2009 and concluded that annual management recommendations developed according to the PFMC's Pacific Coast Salmon FMP and its associated amendments were not likely to jeopardize the continued existence of the SRKW Distinct Population Segment (DPS) or adversely modify its critical habitat. Given new information (since 2009) is available on SRKW and their prey, and potentially the effects of the fisheries on the whales, NMFS has re-initiated ESA consultation on the Council fisheries, and asked for the Council's assistance in assessing the effects of implementing the FMP in 2019 and beyond. In cooperation, the Council appointed a workgroup with membership including representatives from West Coast tribes; the states of California, Oregon, Washington, and Idaho; the PFMC; and NMFS' West Coast Region, Northwest Fisheries Science Center, and Southwest Fisheries Science Center.

The purpose the Council tasked the workgroup with was to reassess the effects of PFMC ocean salmon fisheries on SRKW and if needed, develop a long-term approach that may include proposed conservation measure(s) or management tool(s) that limit PFMC fishery impacts to Chinook salmon prey availability for SRKW relative to implementing the FMP. The need is that the workgroup's findings will inform NMFS' ESA consultation and biological opinion, wherein NMFS will determine whether the fisheries jeopardize the continued existence of SRKWs in light of new information about the whales' dependence on West Coast Chinook salmon stocks.

Specifically, the Workgroup collected and summarized information related to:

- Overlap between PFMC salmon fisheries and SRKW;
- Information the Council's Salmon Technical Team (STT) developed in 2019 regarding which Chinook salmon stocks that are priorities for the whales also contribute to PFMC-area salmon fisheries (see Agenda Item D.8.a, Supplemental STT Report 2 from the Council's 2019 March meeting); and
- Analyses for prior salmon fishery/SRKW evaluations.

The Workgroup was also instructed to recommend (if needed based on the risk assessment) conservation measures or management tools to limit PFMC fishery impacts on Chinook salmon prey availability for SRKW.

In trying to quantify effects on SRKW due to Chinook salmon removals in Council-area ocean salmon fisheries, the Workgroup approached the analysis in four steps:

I) Develop annual indices of adult (age-3+) Chinook salmon abundance by ocean area and three seasonal breakpoints

II) Relate these indices of Chinook salmon abundance to measures of SRKW demographic rates

III) Estimate reductions in Chinook salmon abundance by time and area that are attributable to Council-area ocean salmon fisheries

IV) Estimate the changes in predicted vital rates that the statistical relationships fitted in step II predict for the reductions in abundance estimated in step III.

Details for this methodology and criteria are described in Chapter 5.

The workgroup is focused exclusively on addressing the impacts of PFMC-area ocean salmon fisheries through tools or conservation measures that apply to those fisheries. Considerations of other fisheries or other threats to SRKW are outside the scope of the reinitiated consultation, which is limited to the salmon fisheries as implemented under the FMP. NMFS considers other activities in the action area as part of the environmental baseline in the consultation. In addition, the NMFS West Coast Region and its partners are addressing the broader suite of threats separately.

1.3 NMFS Recovery Plan Guidance

Working with its federal, state, tribal, and local partners, NMFS published a recovery plan for SRKW in January 2008 (NMFS 2008). The plan provides a road map to recovery and there is considerable uncertainty about which threats (prey abundance and quality, noise, and contaminants) may be responsible for the decline in the SRKW population, or which is the most important to address for recovery. The plan lays out an adaptive management approach and a recovery strategy that addresses each of the potential threats based on the best available science. The recovery program outlines links from management actions to an active research program to fill data gaps and a monitoring program to assess effectiveness. Feedback from research and monitoring will provide the information necessary to refine ongoing actions and develop and prioritize new actions. For actions that affect prey abundance, (*e.g.*, salmon), NMFS identified near-term priorities of ongoing restoration efforts for depleted salmon populations in order to:

- Rebuild depleted populations of salmon and other prey to ensure an adequate food base for recovery of SRKWs.
- Support salmon restoration efforts in the region.
- Support regional restoration efforts for other prey species.
- Use NMFS authorities under the ESA and the Magnuson Stevens Act (MSA) to protect prey habitat, regulate harvest, and operate hatcheries.

Healthy SRKW populations are dependent on adequate prey levels. Reductions in prey availability may force SRKWs to spend more time foraging and might lead to reduced reproductive rates or higher mortality rates.

2 STATUS OF THE SPECIES

The SRKW DPS, composed of J, K, and L pods, was listed as endangered under the ESA on November 18, 2005 (70 FR 69903). A 5-year review under the ESA completed in 2016 concluded that SRKWs should remain listed as endangered and includes recent information on the population, threats, and new research results and publications (NMFS 2016).

The limiting factors described in the final recovery plan included reduced prey availability and quality, high levels of contaminants from pollution, and disturbances from vessels and sound (NMFS 2008). This section summarizes the status of SRKWs throughout their range and summarizes information taken largely from the recovery plan (NMFS 2008), recent 5-year review (NMFS 2016), as well as newly available data.

Most of the scientific research conducted on SRKW occurs in inland waters of Washington State and British Columbia. In general, the primary objective of this research is population monitoring or data gathering for behavioral and ecological studies. Research activities are typically conducted between May and October in inland waters and can include aerial surveys, vessel surveys, close approaches, and documentation, and biological sampling.

2.1 Abundance, Productivity, and Trends

SRKW are a long-lived species, sexual maturity occurs at age 10 (review in NMFS (2008)). Females produce a small number of surviving calves (n < 10, but generally fewer) over the course of their reproductive life span (Bain 1990; Olesiuk *et al.* 1990). Compared to Northern Resident killer whales (NRKWs), which are a resident killer whale population with a sympatric geographic distribution ranging from coastal waters of Washington State and British Columbia north to Southeast Alaska, SRKW females appear to have reduced fecundity (Ward *et al.* 2013; Vélez-Espino *et al.* 2014).

Recent aerial imagery corroborates previous notions that SRKWs are thought to have a higher than expected rates of reproductive failure. As can be seen by aerial photogrammetry images collected[1], SRKWs that are pregnant develop pronounced increased width at mid body. The gestation period is about 17 months to 18 months. Robeck *et al.* 2016 estimated a mean gestation of 532 ± 3.1 days and Duffield *et al.* 1995 estimated a mean gestation of 517 ± 20 days. Validation based on pregnant whales in captivity and wild whales that gave birth has shown that aerial images can reliably detect pregnancy by about 9 months. A recent study indicated pregnancy hormones (progesterone and testosterone) can be detected in SRKW feces and have indicated several miscarriages, particularly in late pregnancy (Wasser *et al.* 2017). The fecal hormone data have shown that up to 69 percent of the detected pregnancies do not produce a documented calf (Wasser *et al.* 2017)

Since the early 1970s, annual censuses in the Salish Sea using photo-identification techniques have occurred (Bigg *et al.* 1976; Balcomb *et al.* 1980; Center for Whale Research annual photographic identification catalog, 2019). The surveys are typically performed from May to October, when all

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[1] Presentation on May 23, 2019, to the SRKW Ad Hoc Workgroup: "Photogrammetry to monitor growth and body condition". This work is a collaboration with NOAA SWFSC and SR3 (a non-profit research and animal welfare group based in Seattle). The time series has also had key contributions from the Center for Whale Research on San Juan Island, and the Vancouver Aquarium.

three pods tend to reside near the San Juan Islands, and are considered complete censuses of the entire population. The population was at its lowest known abundance in the early 1970s following live-captures for aquaria display. The abundance since the annual censuses began peaked in 1995 followed by an almost 20 percent decline from 1995-2001 (from 98 whales in 1995 to 81 whales in 2001). In 2014 and 2015, the SRKW population increased from 78 to 81 as a result of multiple successful pregnancies (n = 9) that occurred in 2013 and 2014. At present, the SRKW population has declined to near historically low levels (Figure 2.1.a). As of August 2019, the population is 73 whales (2 calves were born and three whales died since the 2018 census).

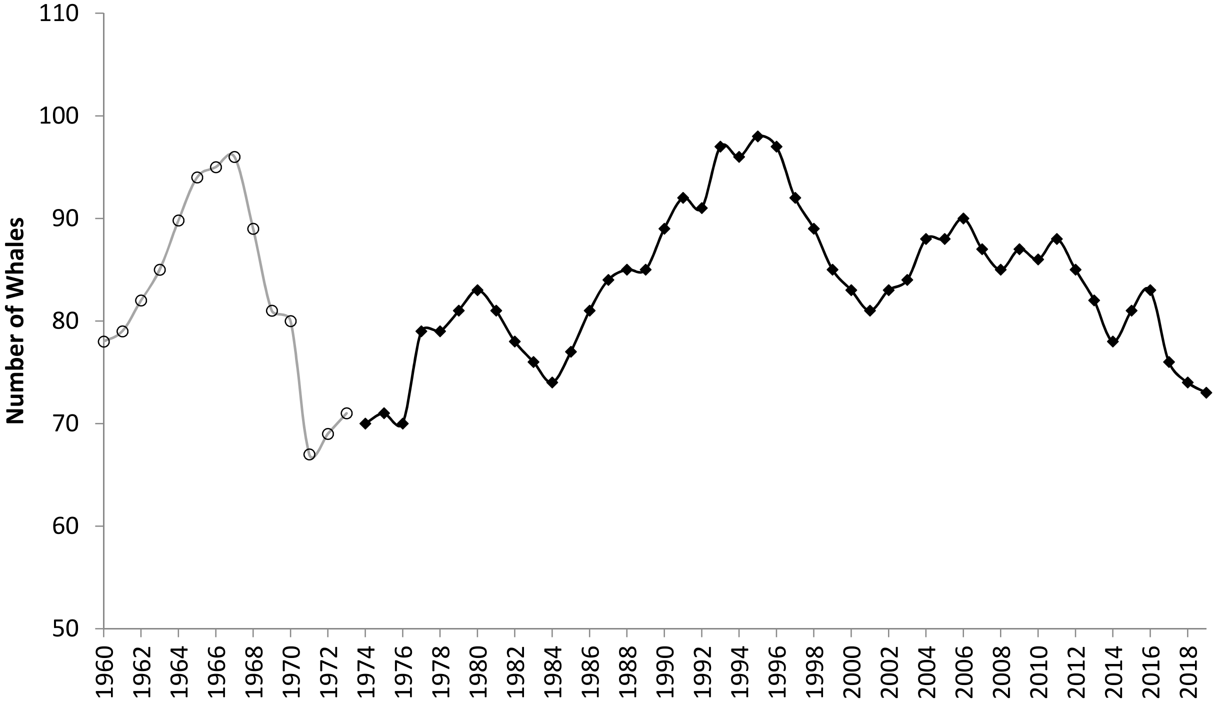

Figure 2.1.a. Population size and trend of Southern Resident killer whales, 1960-2019. Data from 1960-1973 (open circles, gray line) are number projections from the matrix model of Olesiuk *et al.* (1990). Data from 1974-2019 (diamonds, black line) were obtained through photo-identification surveys of the three pods (J, K, and L) in this community and were provided by the Center for Whale Research (unpublished data) and NMFS (2008). Data for these years represent the number of whales present at the end of each calendar year.

There are several demographic factors of the SRKW population that are cause for concern, namely (1) reduced fecundity, (2) a skewed sex ratio toward male births in recent years, (3) a lack of calf production from certain components of the population (K pod, other groups), (4) a small number of adult males acting as sires (Ford *et al.* 2018) and (5) an overall small number of individuals in the population (review in NMFS 2008). Based on an updated pedigree from new genetic data, many of the offspring in recent years were sired by two fathers, meaning that less than 30 individuals make up the effective reproducing portion of the population. Because a small number of males were identified as the fathers of many offspring, a smaller number may be sufficient to support population growth than was previously thought (Ford *et al.* 2011; Ford *et al.* 2018). Inbreeding may be common amongst this small population, with a recent study by Ford *et al.* (2018) finding several offspring resulting from matings between parents and their own offspring.

The fitness effects of this inbreeding remain unclear and is an effort of ongoing research (Ford *et al.* 2018).

The previously published historical abundance of SRKW is 140 animals (NMFS 2008). This estimate (~140) was generated as the number of whales killed or removed for public display in the 1960s and 1970s (summed over all years) added to the remaining population at the time the captures ended. Because of the summed captures over all years, this estimate is likely an upper bound of the population size prior to removals.

The NWFSC continues to evaluate changes in fecundity and mortality rates, and has updated population viability analyses conducted for the 2004 Status Review for Southern Resident Killer Whales and the 2011 science panel review of the effects of salmon fisheries (Krahn *et al.* 2004; Hilborn *et al.* 2012; Ward *et al.* 2013). Following from that work, the data now suggests a downward trend in population growth projected over the next 50 years. As the model projects out over a longer time frame (50 years) there is increased uncertainty around the estimates. The downward trend is in part due to the changing age and sex structure of the population, and will occur more frequently if the fecundity rates are lower (as in 2016) compared to the recent past (2011-2016) as shown in Figure 2.1.b (NMFS 2016).

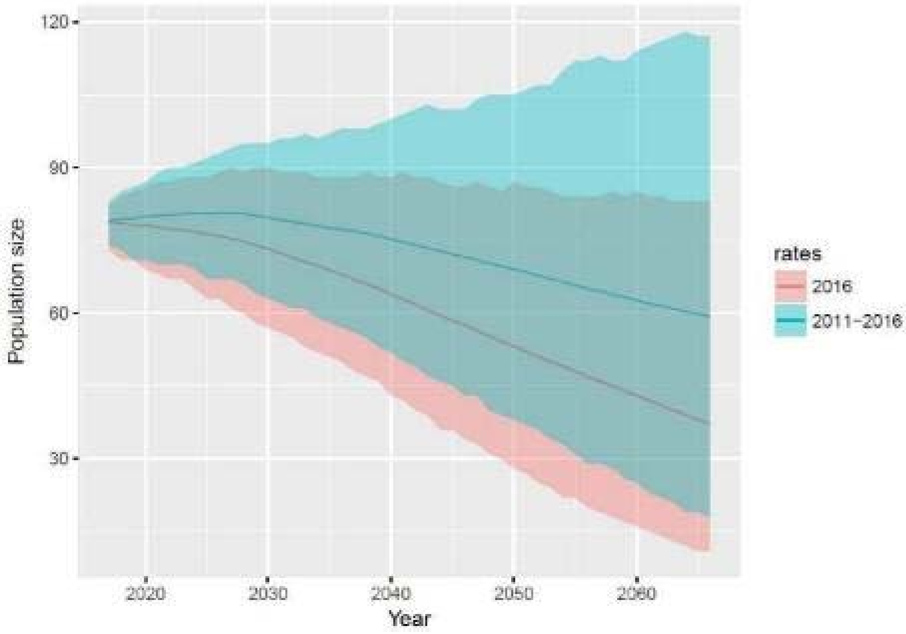

Figure 2.1.b. Southern Resident killer whale population size projections from 2016 to 2066 using 2 scenarios: (1) projections using demographic rates held at 2016 levels, and (2) projections using demographic rates from 2011 to 2016. The pink line represents the projection assuming future rates are similar to those in 2016, whereas the blue represents the scenario with future rates being similar to 2011 to 2016 (Figure 2, NMFS 2016).

Because of this population's small abundance, it is also susceptible to increased risks of demographic stochasticity – randomness in the pattern of births and deaths among individuals in a population. Several other sources of stochasticity can affect small populations and contribute to variance in a population's growth and increased extinction risk. Other sources include environmental stochasticity, or fluctuations in the environment that drive fluctuations in birth and death rates, and demographic heterogeneity, or variation in birth or death rates of individuals because of differences in their individual fitness (including sexual determinations). In combination, these and other sources of random variation combine to amplify the probability of extinction, known as the extinction vortex (Gilpin and Michael 1986; Fagan and Holmes 2006; Melbourne and Hastings 2008). The larger the population size, the greater the buffer against stochastic events and genetic risks. In light of the current small population size and declining status, these conditions reinforce the need to promote immediate population growth.

Population growth is also important because of the influence of demographic and individual heterogeneity on a population's long-term viability. Population-wide distribution of lifetime reproductive success can be highly variable, such that some individuals produce more offspring than others to subsequent generations, and male variance in reproductive success can be greater than that of females (*e.g.* Clutton-Brock 1988; Hochachka 2006). For long-lived vertebrates such as killer whales, some females in the population might contribute less than the number of offspring required to maintain a constant population size (n = 2), while others might produce more offspring. The smaller the population, the more weight an individual's reproductive success has on the population's growth or decline (Coulson *et al.* 2006). For example, from 2010 through July 2019, only 15 of the 28 reproductive aged females successfully reproduced, resulting in 16 calves. There were an additional 10 documented non-viable calves, and likely more undocumented, born during this period (CWR unpubl. data). This further illustrates the risk of demographic stochasticity for a small population like SRKWs – the smaller a population, the greater the chance that random variation will result in too few successful individuals to maintain the population.

2.2 Geographic Range and Distribution

On September 19, 2019 NMFS proposed to revise the critical habitat designation for the SRKW DPS under the ESA by designating six new areas along the U.S. West Coast (84 FR 49214). Specific new areas proposed along the U.S. West Coast include 15,626.6 square miles (mi\2\) (40,472.7 square kilometers (km\2\)) of marine waters between the 6.1-meter (m) (20 feet (ft)) depth contour and the 200-m (656.2 ft) depth contour from the U.S. international border with Canada south to Point Sur, California. The proposed rule to revise critical habitat designation was based on new information about the SRKW's habitat use along the coast. This new information on the whales' distribution and habitat use was also used in this risk assessment and is summarized below.

SRKWs occur throughout the coastal waters off Washington, Oregon, and Vancouver Island and are known to travel as far south as central California and as far north as Southeast Alaska (NMFS 2008; Carretta *et al.* 2019) (Figure 2.2.a). SRKW are highly mobile and can travel up to approximately 86 miles in a single day (Erickson 1978; Baird 2000), with seasonal movements likely tied to the migration of their primary prey, salmon. During the spring, summer, and fall months, SRKWs have typically spent a substantial amount of time in the inland waterways of the Strait of Georgia, Strait of Juan de Fuca, and Puget Sound (Bigg 1982; Ford *et al.* 2000; Krahn *et al.* 2002; Hauser *et al.* 2007). During fall and early winter, SRKWs, and J pod in particular, expand their routine movements into Puget Sound, likely to take advantage of chum, coho, and Chinook salmon runs (Osborne 1999; Hanson *et al.* 2010; Ford *et al.* 2016). Although seasonal movements are generally predictable, there can be large inter-annual variability in arrival time and days present in inland waters from spring through fall, with late arrivals and fewer days present in recent years (Hanson and Emmons 2010; The Whale Museum unpubl. data).

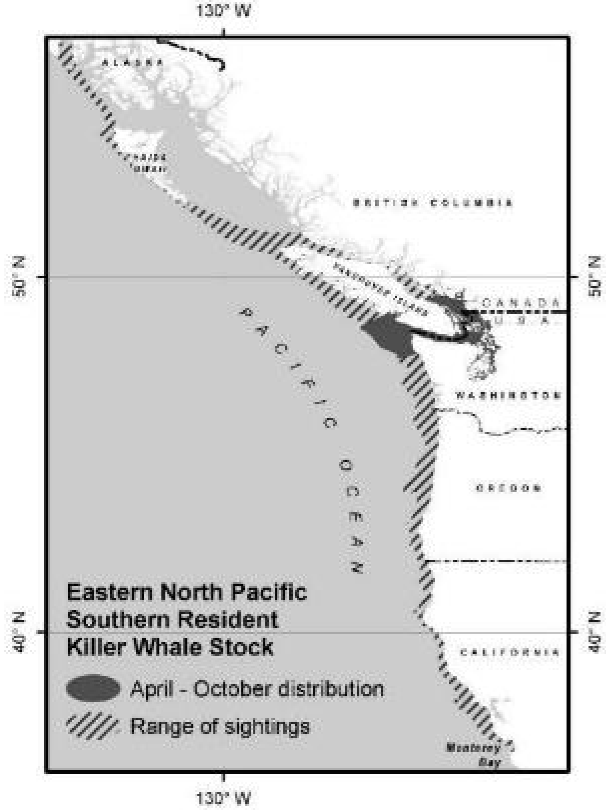

Figure 2.2.a. Approximate April – October distribution of Southern Resident killer whales (shaded area) and range of sightings (diagonal lines) (reprinted from Carretta *et al.* 2019).

SRKWs distribution and forging areas in winter months is less known. Research is on-going to address this. Hanson *et al.* (2017) used remotely deployable tags and acoustic recorders to further understand SRKW movements and occurrence patterns. Hanson *et al.* (2018) integrated opportunistic visual sightings with results from a state-space movement model to fill in the detection gaps in the acoustic detections of SRKWs in coastal waters over a 4-year period when satellite tags were not deployed. From 2012-2016 they deployed satellite-linked tags on eight male SRKW (three tags on J pod members, two on K pod, and three on L pod) in Puget Sound or in the coastal waters of Washington and Oregon (Table 2.2.a). These telemetry tags transmitted multiple locations per day and were used to assess winter movements and occurrences of SRKW (identified as a priority area of research in Hilborn *et al.* 2012). Additionally, passive acoustic recorders were deployed in areas thought to be of frequent use by SRKWs to assess their seasonal uses of these areas via the recording of stereotypic calls of the SRKW. The recorders were deployed off the coasts of California, Oregon and Washington in most years since 2006 (Figure 2.2.b; Hanson *et al.* 2013). The acoustic monitoring detected SRKW 131 times at up to 7 locations from 2006-2011 (Hanson *et al.* 2013). The number of sites off the Washington coast was increased from 7 to 17 in the fall of 2014 to better understand the residency of SRKWs in this area (Figure 2.2.c; Hanson *et al.* 2017).

Table 2.2.a. Satellite-linked tags deployed on Southern resident killer whales 2012-2016 (Hanson et al. 2018).

Whale ID	Pod association	Date of tagging	Duration of signal contact (days)
J26	J	20 Feb. 2012	3
L87	J	26 Dec. 2013	31
J27	J	28 Dec. 2014	49
K25	K	29 Dec. 2012	96
L88	L	8 Mar. 2013	8
L84	L	17 Feb. 2015	93
K33	K	31 Dec. 2015	48
L95	L	23 Feb. 2016	3

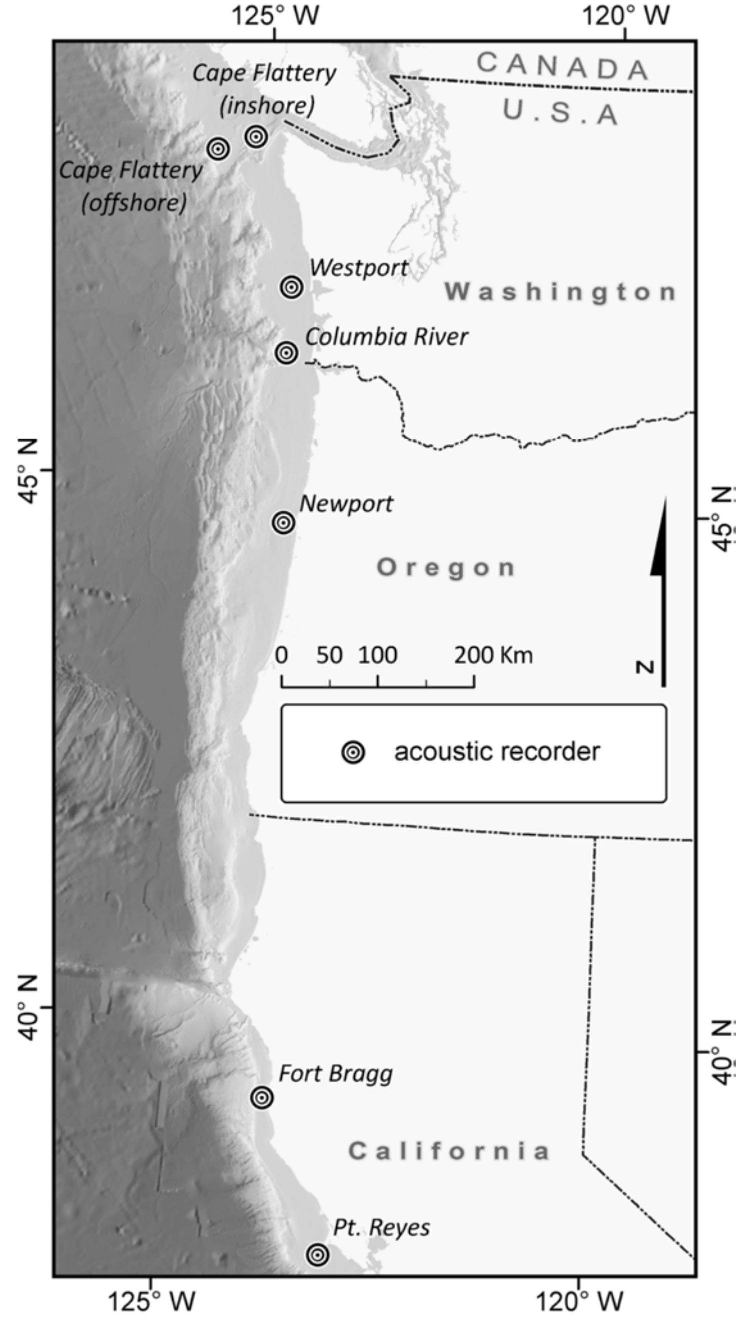

Figure 2.2.b. Deployment locations of acoustic recorders on the U.S. west coast from 2006 to 2011 (Hanson *et al.* 2013).

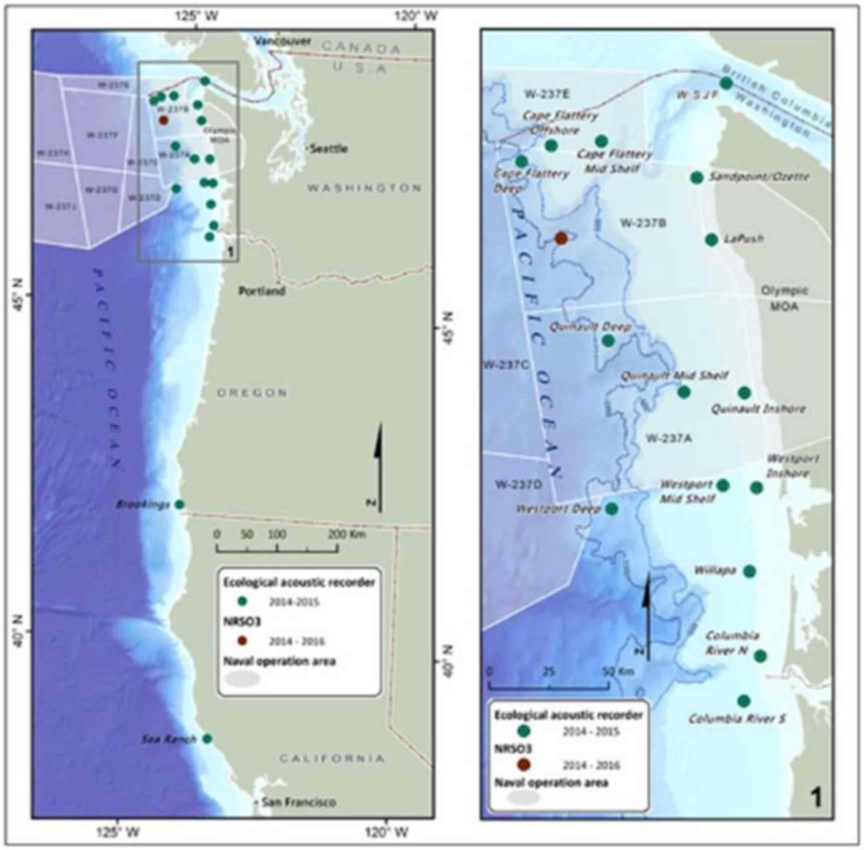

Figure 2.2.c. Locations of passive acoustic recorders deployed beginning in the fall of 2014 (Hanson *et al.* 2017).

The satellite tags resulted in 323 days of monitoring with deployment durations from late December to mid-May. The winter locations of the tagged whales included inland and coastal waters. The inland waters range occurs across the entire Salish Sea, from the northern end of the Strait of Georgia and Puget Sound, and coastal waters from central west coast of Vancouver Island, British Columbia to northern California. J pod had high use areas in the northern Strait of Georgia and the west entrance to the Strait of Juan de Fuca. K/L pods occurred almost exclusively on the continental shelf during December to mid-May, primarily on the Washington coast, with a continuous high use area between Grays Harbor and the Columbia River and off Westport (Hanson *et al.* 2017, 2018). Approximately 95 percent of the SRKW locations were within 34 km of the shore and 50 percent of these were within 10 km of the coast. Only 5 percent of locations were greater than 34 km away from the coast, but no locations exceeded 75 km. Most locations were in waters less than 100m in depth.

Between 2011 and 2016, the SRKWs were acoustically detected 246 times (Hanson *et al.* 2018). There were acoustic detections off Washington coast in all months of the year, with greater than 2.4 detections per month from January through June with a peak of 4.7 detections per month in both March and April (Hanson *et al.* 2017), indicating that the SRKW may be present in Washington coastal waters at nearly any time of year, and in other coastal waters more often than previously believed. High use areas for the SRKW in winter are primarily located in three areas 1) the Washington coast, particularly between Grays Harbor and the mouth of the Columbia River (primarily for K/L pods); 2) the west entrance to the Strait of Juan de Fuca (primarily for J pod); and 3) the northern Strait of Georgia (primarily for J pod). It is important to note that this study was designed to assess spatial use off Washington coast and thus the effort was higher (*i.e.* the number of recorders increased from 7 to 17 in this area, see Figure 2.2.c) compared to off Oregon and California.

In a recent study, researchers collected data using an autonomous acoustic recorder deployed at Swiftsure Bank from August 2009 to July 2011 to assess how this area is used by Northern Resident and Southern Resident killer whales as shown in Figure 2.2.d (Rivera *et al.* 2019).

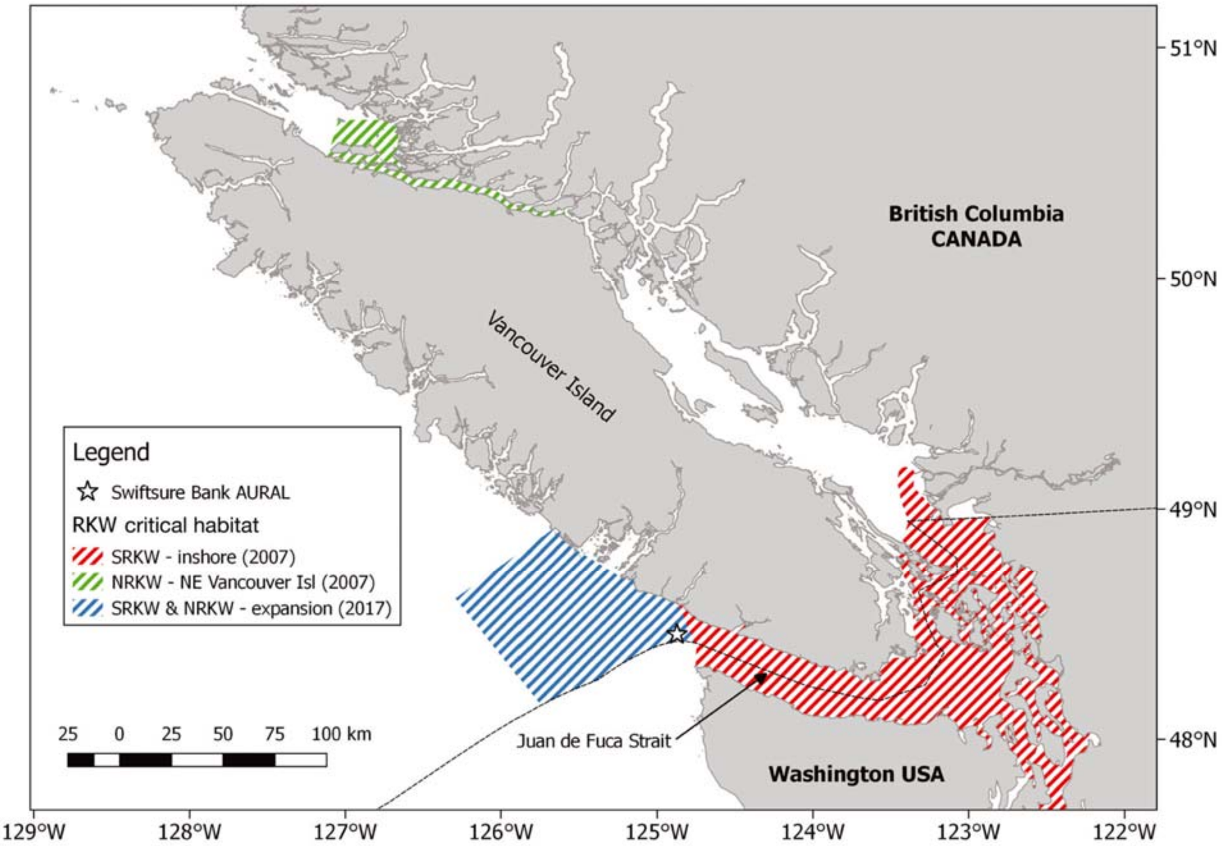

Figure 2.2.d. Swiftsure Bank study site off the coast of British Columbia, Canada (Riera *et al.* 2019).

SRKW were detected on 163 days with 175 encounters (see Figure 2.2.e for number of days of acoustic detections for each month). All three pods were detected at least once per month except for J pod in January and November and L pod in March. K and L pods were heard more often during the summer (87 percent of calls and 89 percent of calls, respectively, between May and September). J pod was heard most often during winter and spring (76 percent of calls during December and February through May; Riera *et al.* 2019). K pod had the longest encounters in June, with 87 percent of encounters longer than 2 hours occurring between June and September. L pod had the longest encounters in May, with 79 percent of encounters longer than two hours occurring during the summer (May through September). The longest J pod encounters were during winter, with 72 percent of encounters longer than 2 hours occurring between December and May (Riera *et al.* 2019).

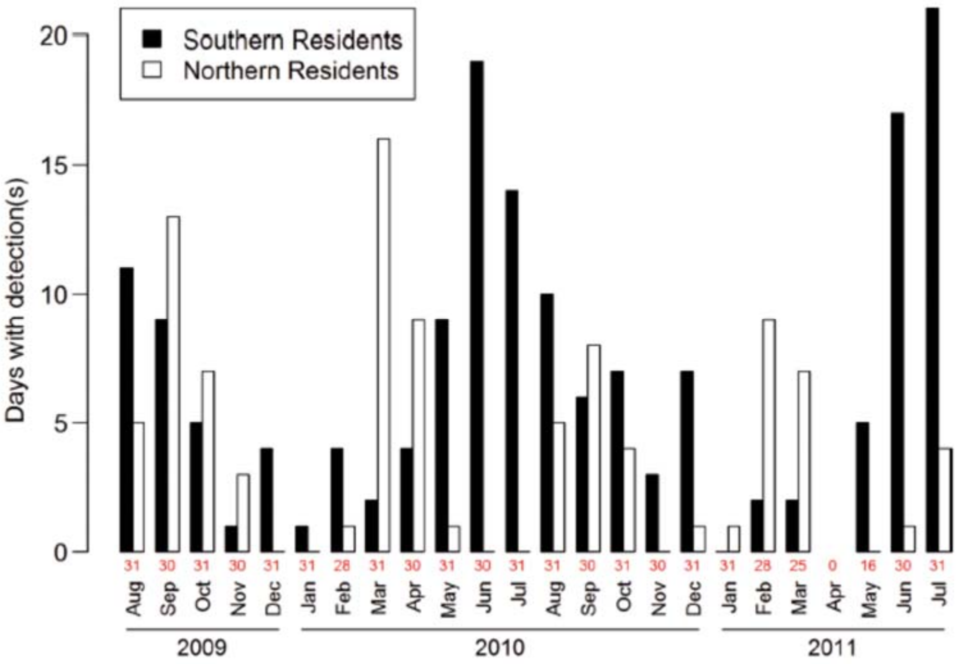

Figure 2.2.e. Number of days with acoustic detections of SRKWs at Swiftsure Bank from August 2009 – July 2011). Red numbers indicate days of effort. (Riera *et al.* 2019).

2.3 Limiting Factors and Threats

Several factors identified in the recovery plan for SRKW may be limiting recovery. These are quantity and quality of prey, toxic chemicals that accumulate in top predators, impacts from sound and vessels. Oil spills are also a risk factor. It is likely that multiple threats are acting together to impact SRKWs. Modeling exercises have attempted to identify which threats are most significant to survival and recovery (*e.g.* Lacy *et al.* 2017) and available data suggest that all of the threats are potential limiting factors (NMFS 2008).

2.3.1 Quantity and Quality of Prey

SRKWs have been documented to consume a variety of fish species (22 species) and one species of squid (Ford *et al.* 1998; Ford *et al.* 2000; Ford and Ellis 2006; Hanson *et al.* 2010; Ford *et al.* 2016), but salmon are identified as their primary prey. SRKWs are the subject of ongoing research, the majority of which has occurred in inland waters of Washington State and British Columbia, Canada during summer months and includes direct observation, scale and tissue sampling of prey remains, and fecal sampling. The diet data suggest that SRKWs are consuming mostly larger (*i.e.*, generally age 3 and up) Chinook salmon (Ford and Ellis 2006). Chinook salmon is their primary prey despite the much lower abundance in comparison to other salmonids in some areas and during certain time periods (Ford and Ellis 2006). Factors of potential importance include the species' large size, high fat and energy content, and year-round occurrence in the SRKWs' geographic range. Chinook salmon have the highest value of total energy content compared to other salmonids because of their larger body size and higher energy density (kilocalorie/kilogram (kcal/kg)) (O'Neill *et al.* 2014). For example, in order for a SRKW to obtain the total energy value of one adult Chinook salmon, they would need to consume approximately 2.7 coho, 3.1 chum, 3.1

sockeye, or 6.4 pink salmon (O'Neill *et al.* 2014). Research suggests that SRKWs are capable of detecting, localizing, and recognizing Chinook salmon through their ability to distinguish Chinook echo structure as different from other salmon (Au *et al.* 2010).

May - September

Scale and tissue sampling from May to September in inland waters of Washington and British Columbia, Canada indicate that the SRKW's diet consists of a high percentage of Chinook salmon (monthly proportions as high as >90 percent) (Hanson *et al.* 2010; Ford *et al.* 2016). Genetic analysis of the Hanson *et al.* (2010) samples from 2006-2010 indicate that when SRKW are in inland waters from May to September, they primarily consume Chinook stocks that originate from the Fraser River (80–90 percent of the diet in the Strait of Juan de Fuca and San Juan Islands; including Upper Fraser, Mid Fraser, Lower Fraser, North Thompson, South Thompson and Lower Thompson), and to a lesser extent consume stocks from Puget Sound (North and South Puget Sound) and Central British Columbia Coast and West and East Vancouver Island. This is not unexpected as all of these stocks are returning to streams proximal to these inland waters during this timeframe.

DNA quantification methods are also used to estimate the proportion of different prey species in the diet from fecal samples (Deagle *et al.* 2005). Recently, Ford *et al.* (2016) confirmed the importance of Chinook salmon to SRKWs in the early to mid-summer months (May-August) using DNA sequencing from SRKW feces collected in inland waters of Washington and British Columbia. Salmon and steelhead made up to 98 percent of the inferred diet, of which almost 80 percent were Chinook salmon. Coho salmon and steelhead are also found in the diet in inland waters of Washington and British Columbia in spring and fall months when Chinook salmon are less abundant. Specifically, coho salmon contribute to over 40 percent of the diet in September in inland waters, which is evidence of prey shifting at the end of summer towards coho salmon (Ford *et al.* 1998; Ford and Ellis 2006; Hanson *et al.* 2010; Ford *et al.* 2016). Less than 3 percent each of chum salmon, sockeye salmon, and steelhead were observed in fecal DNA samples collected in the summer months (May through September) in inland waters.

October - December

Prey remains and fecal samples collected in inland waters during October through December indicate Chinook and chum salmon are primary contributors of the whale's diet during this time (NWFSC unpublished data; Figure 2.3.a). Diet data for coastal waters is limited.

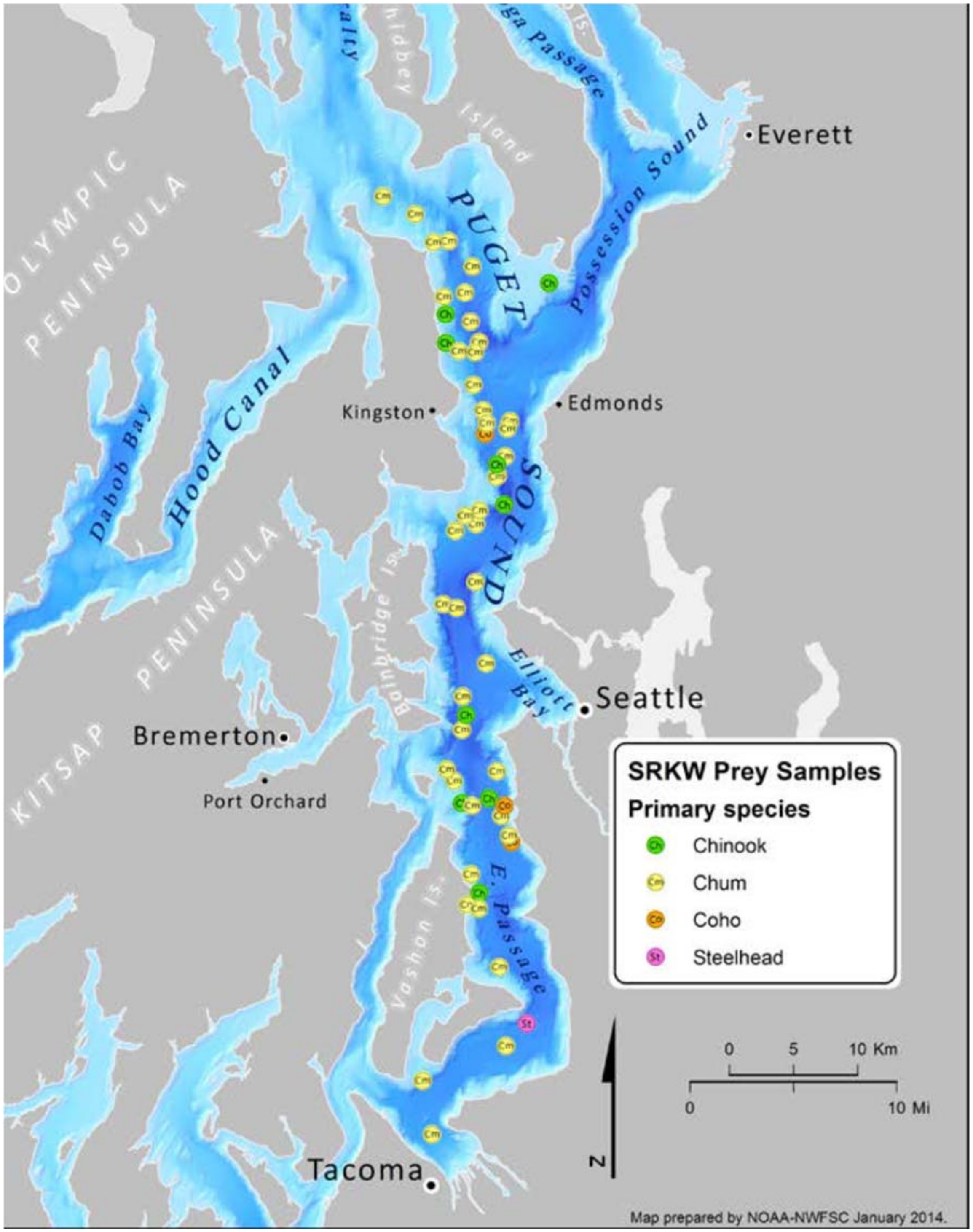

Figure 2.3.a Map of locations of SRKW predation events by prey species in Puget Sound between October and December (NWFSC unpubl. data).

January – April

Observations of SRKWs overlapping with salmon runs (Wiles 2004; Zamon *et al.* 2007) and collection of prey and fecal samples have also occurred in coastal waters in the winter and spring months. Although fewer predation events have been observed and less fecal samples collected in coastal waters, recent data indicate that salmon, and Chinook salmon in particular, remains an important dietary component when the SRKWs occur in outer coastal waters during these timeframes. Prior to 2013, only three prey samples for SRKW on the U.S. outer coast had been collected (Hanson *et al.* in prep). From 2013 to 2016, satellite tags were used to locate and follow the whales to obtain predation and fecal samples. A total of 55 samples were collected from northern California to northern Washington (Figure 2.3.b). Results of the 55 available prey samples indicate that, as is the case in inland waters, Chinook are the primary species detected in diet samples on the outer coast, although steelhead, chum, lingcod, and halibut were also detected in samples.

The occurrence of K and L pods off the Columbia River in March suggests the importance of Columbia River spring runs of Chinook salmon in their diet (Hanson *et al.* 2013). Chinook genetic stock identification from samples collected in winter and spring in coastal waters from California through Washington included 12 U.S. west coast stocks, and showed that over half the Chinook salmon consumed originated in the Columbia River (Ward, May 23, 2019; Workgroup Agenda Item B.3; Figure 2.3.b). Columbia River, Central Valley, Puget Sound, and Fraser River Chinook salmon collectively comprised over 90 percent of the 55 diet samples collected for SRKW's in coastal areas (Ward, May 23, 2019; Workgroup Agenda Item B.3).

As noted, most of the Chinook prey samples opportunistically collected in coastal waters were determined to have originated from the Columbia River basin, including Lower Columbia Spring, Middle Columbia Tule, and Upper Columbia Summer/Fall. In general, we would expect to find these stocks given the diet sample locations (Figure 2.3.1.b) However, the Chinook stocks included fish from as far north as the Taku River (Alaska and British Columbia stocks) and as far south as the Central Valley California (Ward, May 23, 2019; Workgroup Agenda Item B.3).

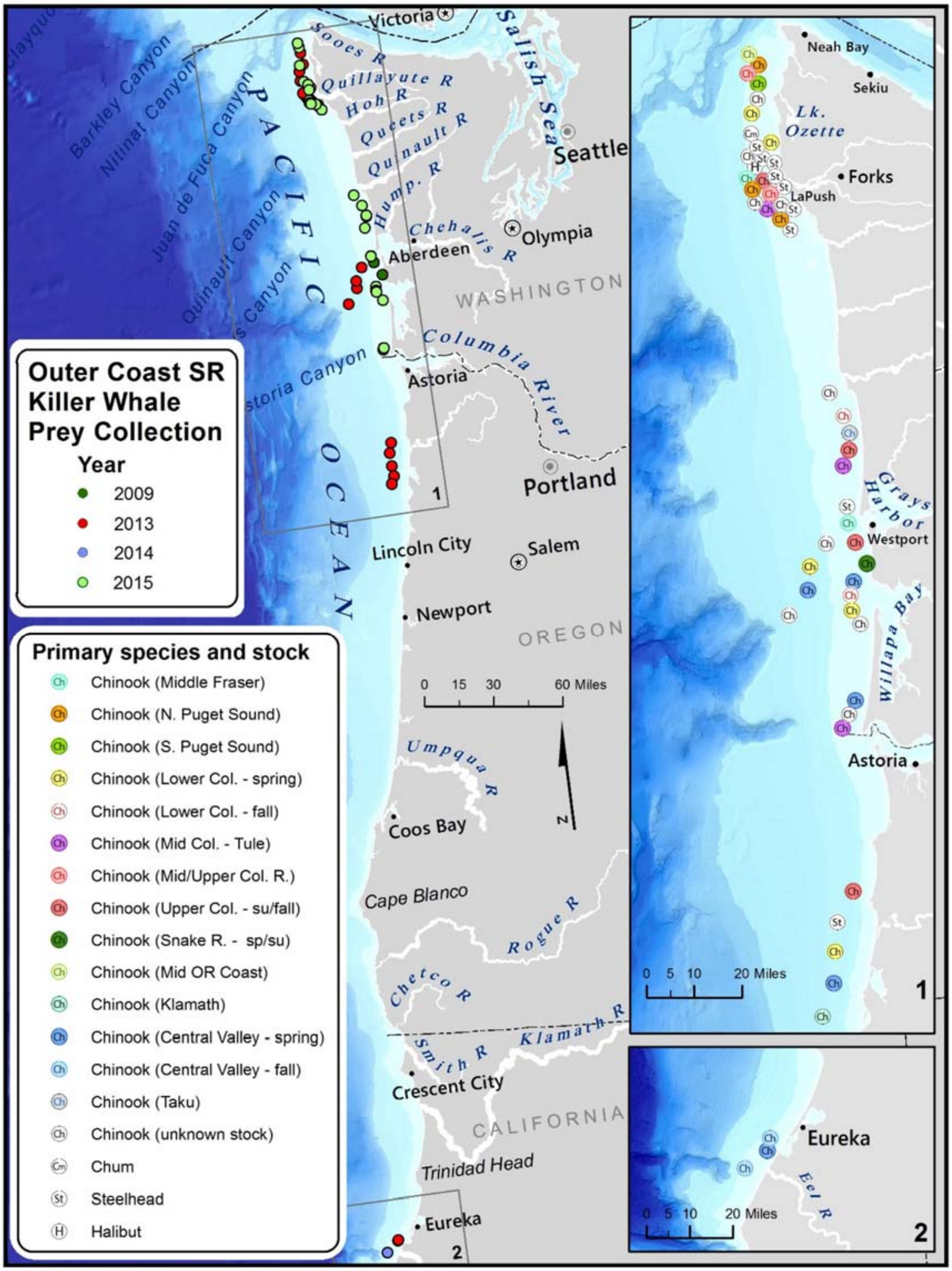

Figure 2.3.b. Location and species for scale/tissue samples collected from Southern Resident killer whale predation events in outer coastal waters[2]

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[2] Ward presentation to the Workgroup on May 23, 2019; Agenda Item B.3

In general, over the past decade, some Chinook salmon stocks within the range of the SRKWs have had relatively high abundance (*e.g.* Washington (WA)/Oregon (OR) coastal stocks, some Columbia River stocks) compared to the previous decade, whereas other stocks originating in the more northern and southern ends of the whales' range (*e.g.* most Fraser stocks, Northern and Central British Columbia (B.C.) stocks, Georgia Strait, Puget Sound, and Central Valley) have declined. There are many factors that affect the abundance, productivity, spatial structure, and diversity of Chinook salmon and thus affect prey availability for the whales. Human impacts and limiting factors come from multiple sources, including hydropower development, habitat degradation, hatchery effects, fishery management decisions, and ecological factors, including predation and environmental variability. Changing ocean conditions driven by climate change have influenced ocean survival and distribution of Chinook and other Pacific salmon, affecting the prey available to SRKWs.

In an effort to prioritize recovery efforts to increase the whales' prey base, NMFS and WDFW developed a report identifying the Chinook salmon stocks thought to be of most importance to the health of the SRKW populations along the West Coast (NOAA and WDFW 2018)[3]. Scientists and managers from the U.S. and Canada reviewed the model at a workshop sponsored by the National Fish and Wildlife Foundation (NFWF). The priority stock report was created using observations of Chinook salmon stocks found in scat and prey scale/tissue samples, and by estimating the spatial and temporal overlap with Chinook salmon stocks ranging from Southeast Alaska (SEAK) to California (CA). The report gave higher priority to salmon runs that support the SRKWs during times of the year when the SRKWs' body condition is more likely reduced (October through May) and when Chinook salmon may be less available, such as in winter months. The analysis also placed higher priority on Chinook populations observed in the SRKW diet samples as being more important to the health of the whales. The Workgroup reviewed the priority prey list and modified it to reflect error checking and also revised salmon distributions that were suggested for a few stocks by workgroup members. Table 2.3.a. is a summary of those stock descriptions. However, because the list was developed to help prioritize salmon recovery actions and was not developed to describe or assess prey availability along the coast, the Workgroup decided to instead develop a quantitative method to assess available abundances of Chinook stocks, which is described in Section 5.1 of this document.

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[3] https://www.westcoast.fisheries.noaa.gov/publications/protected_species/marine_mammals/killer_whales/recovery/srkw_priority_chinook_stocks_conceptual_model_report___list_22june2018.pdf

Table 2.3.a. Summary of the priority Chinook salmon stocks (adapted from NOAA and WDFW (2018), updated based on error checking and revised distribution estimates provided by workgroup members). This list was developed to assist in prioritizing actions (*e.g.*, habitat restoration and hatchery production) to increase critical prey for the whales. The Workgroup decided not to use this list to inform fisheries impacts and established its own methodology to estimate prey availability.

Priority	ESU/Stock Group	Run Type	Rivers or Stocks in Group
1	North Puget Sound	Fall	Nooksack, Elwha, Dungeness, Skagit, Stillaguamish, Snohomish, Nisqually, Puyallup, Green, Duwamish, Deschutes, Hood Canal Systems
	South Puget Sound		
2	Lower Columbia Strait of Georgia	Fall	Fall Tules and Fall Brights (Cowlitz, Kalama, Clackamas, Lewis, others), Lower Strait (Cowichan, Nanaimo), Upper Strait (Klinaklini, Wakeman, others), Fraser (Harrison)
3	Upper Columbia & Snake	Fall	Upriver Brights
4	Middle Columbia	Fall	Fall Brights
5	Fraser	Spring	Spring 1.3 (Upper Pitt, Birkenhead; Mid & Upper Fraser; North and South Thompson) and Spring 1.2 (Thompson, Louis Creek, Bessette Creek);
	Lower Columbia	Spring	Lewis, Cowlitz, Kalama, Big White Salmon
	Snake River	Spring/summer	Snake, Salmon, Clearwater
	Northern Puget Sound	Spring	Nooksack, Elwha, Dungeness, Skagit (Stillaguamish, Snohomish)
	Washington Coast	Spring	Hoh, Queets, Quillayute, Grays Harbor
6	Washington Coast	Fall	Hoh, Queets, Quillayute, Grays Harbor
7	Central Valley	Fall and late Fall	Sacramento, San Joaquin
		Spring	Sacramento and tributaries

Priority	ESU/Stock Group	Run Type	Rivers or Stocks in Group
8	Middle/Upper Columbia	Spring/Summer	Columbia, Yakima, Wenatchee, Methow, Okanagan
9	Fraser	Summer	Summer 0.3 (South Thompson, Lower Fraser, Shuswap, Adams, Little River, Maria Slough) and Summer 1.3 (Nechako, Chilko, Quesnel, Clearwater River)
10	Klamath River	Fall and Spring	Upper Klamath, and Trinity
11	Upper Willamette	Spring	Willamette
	South Puget Sound	Spring	Nisqually, Puyallup, Green, Duwamish, Deschutes, Hood Canal systems
13	West Coast Vancouver Island	Fall	Robertson Creek, WCVI Wild
14	North/Central Oregon Coast	Fall	Northern (Siuslaw, Nehalem, Siletz) and Central (Coos, Elk, Coquille, Umpqua)
15	Southern OR & Northern CA Coastal	Fall and Spring	Rogue, Chetco, Smith, Lower Klamath, Mad, Eel, Russian
16	Central Valley	Winter	Sacramento and tributaries

Currently, hatchery production is a significant component of the salmon prey base returning to watersheds within the range of SRKWs (Barnett-Johnson *et al.* 2007; NMFS 2008). The release of hatchery fish has not been identified as a threat to the survival or persistence of SRKWs. It is possible that hatchery produced fish may benefit this endangered population of whales by enhancing prey availability to SRKWs and hatchery fish often contribute significantly to the salmon stocks consumed (Hanson *et al.* 2010). Currently, hatchery fish play a mitigation role of helping sustain Chinook salmon numbers while other, longer term, recovery actions for natural fish are underway.

2.3.2 Nutritional Limitation and Body Condition

When prey is scarce or in low density, SRKWs likely spend more time foraging than when prey is plentiful or in high density. Increased energy expenditure and prey limitation can cause poor body condition and nutritional stress. Nutritional stress is the condition of being unable to acquire adequate energy and nutrients from prey resources and as a chronic condition, can lead to reduced body size of individuals and to lower reproductive or survival rates in a population (Trites and Donnelly 2003). During periods of nutritional stress and poor body condition, cetaceans lose adipose tissue behind the cranium, displaying a condition known as "peanut-head" in extreme cases (Pettis *et al.* 2004; Bradford *et al.* 2012; Joblon *et al.* 2014). Between 1994 and 2008, 13 SRKWs were observed from boats to have a pronounced "peanut-head"; and all but two subsequently died (Durban *et al.* 2009; Center for Whale Research unpublished data). None of the whales that died were subsequently recovered, and therefore definitive cause of death could not be identified. Both females and males across a range of ages were found in poor body condition.

Since 2008, NOAA's Southwest Fishery Science Center (SWFSC) and SR^3, a response rehabilitation and research center, have used aerial photogrammetry to assess the body condition and health of SRKWs, initially in collaboration with the Center for Whale Research and with the Vancouver Aquarium. Aerial photogrammetry studies have provided finer resolution for detecting poor condition, even before it manifests in "peanut-head" that is observable from boats. Annual aerial surveys of the population from 2013-2017 (with exception of 2014) have detected declines in condition before the death of seven SRKWs (L52 and J8 as reported in Fearnbach *et al.* (2018); J14, J2, J28, J54, and J52 as reported in Durban *et al.* (2017)), including five of the six most recent mortalities (Trites and Rosen 2018). These data have provided evidence of a general decline in SRKW body condition since 2008, and documented members of J pod being in poorer body condition in May compared to September of the previous year (at least in 2016 and 2017) (Trites and Rosen 2018). Other pods could not be reliably photographed in both seasonal periods.

Previous scientific review investigating nutritional stress as a cause of poor body condition for SRKWs concluded "Unless a large fraction of the population experienced poor condition in a particular year, and there was ancillary information suggesting a shortage of prey in that same year, malnutrition remains only one of several possible causes of poor condition" (Hilborn *et al.* 2012). Body condition in whales can be influenced by a number of factors, including prey availability or limitation, increased energy demands, disease, physiological or life history status, and variability over seasons or across years. Body condition data collected to date has documented declines in condition for some animals in some pods and these occurrences have been scattered across demographic and social groups (Fearnbach *et al.* 2018).

It is possible that poor nutrition could contribute to mortality through a variety of mechanisms. To exhibit how this is possible, we reference studies that have demonstrated the effects of energetic stress (caused by incremental increases in energy expenditures or incremental reductions in available energy) on adult females and juveniles, which have been studied extensively (*e.g.*, adult females: Gamel *et al.* (2005), Schaefer (1996), Daan *et al.* (1996), juveniles: Trites and Donnelly (2003)). Small, incremental increases in energy demands should have the same effect on an animal's energy budget as small, incremental reductions in available energy, such as one would expect from reductions in prey. Ford and Ellis (2006) report that SRKWs engage in prey sharing about 76 percent of the time. Prey sharing presumably would distribute more evenly the effects of prey limitation across individuals of the population than would otherwise be the case (*i.e.,* if the most successful foragers did not share with other individuals).

2.3.3 Toxic Chemicals

Because the PFMC has little to no control over toxic chemicals thereby precluding a pathway for effects of toxic chemicals on SRKWs due to the PFMC salmon fisheries, we only briefly describe this threat (see NMFS 2008 for more information). Various adverse health effects in humans, laboratory animals, and wildlife have been associated with exposures to persistent pollutants. These pollutants have the ability to cause endocrine disruption, reproductive disruption or failure, immunotoxicity, neurotoxicity, neurobehavioral disruption, and cancer (Reijnders 1986; Subramanian *et al.* 1987; de Swart *et al.* 1996; Bonefeld-Jørgensen *et al.* 2001; Reddy *et al.* 2001; Schwacke *et al.* 2002; Darnerud 2003; Legler and Brouwer 2003; Viberg *et al.* 2003; Ylitalo *et al.* 2005; Fonnum *et al.* 2006; Darnerud 2008; Legler 2008). SRKWs are exposed to a mixture of pollutants, some of which may interact synergistically and enhance toxicity, influencing their health. Relatively high levels of these pollutants have been measured in blubber biopsy samples from SRKWs compared to other resident killer whales in the North Pacific (Ross *et al.* 2000; Krahn *et al.* 2007; Krahn *et al.* 2009; Lawson *et al.* in prep), and more recently, these pollutants were measured in fecal samples collected from SRKWs providing another potential opportunity to evaluate exposure to these pollutants (Lundin *et al.* 2016a; Lundin *et al.* 2016b).

SRKWs are exposed to persistent pollutants primarily through their diet. For example, Chinook salmon contain higher levels of some persistent pollutants than other salmon species, but only limited information is available for pollutant levels in Chinook salmon (Krahn *et al.* 2007; O'Neill and West 2009; Veldhoen *et al.* 2010; Mongillo *et al.* 2016). These harmful pollutants, through consumption of prey species that contain these pollutants, are stored in the SRKW's blubber and can later be released; when the pollutants are released, they are redistributed to other tissues when the SRKWs metabolize the blubber in response to food shortages or reduced acquisition of food energy that could occur for a variety of other reasons. The release of pollutants can also occur during gestation or lactation. Once the pollutants mobilize in to circulation, they have the potential to cause a toxic response. Therefore, nutritional stress from reduced Chinook salmon populations may act synergistically with high pollutant levels in SRKWs and result in adverse health effects.

2.3.4 Disturbance from Vessels and Sound

Because the Council only tasked the Workgroup to assess impacts from the PFMC ocean salmon fisheries on prey availability, we did not assess the impacts of fishing vessels on the whales. However, here we provide a general description of this threat. Vessels have the potential to affect SRKWs through the physical presence and activity of the vessel, increased underwater sound levels generated by boat engines, or a combination of these factors. Vessel strikes are rare, but do occur and can result in injury or mortality (Gaydos and Raverty 2007). In addition to vessels, underwater sound can be generated by a variety of other human activities, such as dredging, drilling, construction, seismic testing, and sonar (Richardson *et al.* 1995; Gordon and Moscrop 1996; National Research Council 2003). Impacts from these sources can range from serious injury and mortality to changes in behavior. In other cetaceans, hormonal changes indicative of stress have been recorded in response to intense sound exposure (Romano *et al.* 2003). Chronic stress is known to induce harmful physiological conditions including lowered immune function, in terrestrial mammals and likely does so in cetaceans (Gordon and Moscrop 1996).

Killer whales rely on their highly developed acoustic sensory system for navigating, locating prey, and communicating with other individuals. While in inland waters of Washington and British Columbia, SRKWs are the principal target species for the commercial whale watch industry (Hoyt 2001; O'Connor *et al.* 2009) and encounter a variety of other vessels in their urban environment (*e.g.*, recreational, fishing, ferries, military, shipping). Several main threats from vessels include direct vessel strikes, the masking of echolocation and communication signals by anthropogenic sound, and behavioral changes (NMFS 2008). There is a growing body of evidence documenting effects from vessels on small cetaceans and other marine mammals. Research has shown that SRKWs spend more time traveling and performing surface active behaviors and less time foraging in the presence of all vessel types, including kayaks, and that noise from motoring vessels up to 400 meters away has the potential to affect the echolocation abilities of foraging whales (Holt 2008; Lusseau *et al.* 2009; Noren *et al.* 2009; Williams *et al.* 2010). Individual energy balance may be impacted when vessels are present because of the combined increase in energetic costs resulting from changes in whale activity with the decrease in prey consumption resulting from reduced foraging opportunities (Williams *et al.* 2006; Lusseau *et al.* 2009; Noren *et al.* 2009; Noren *et al.* 2012).

At the time of the SRKWs' listing under the ESA, NMFS reviewed existing protections for the whales and developed recovery actions, including vessel regulations, to address the threat of vessels to SRKWs. NMFS concluded it was necessary and advisable to adopt regulations to protect SRKWs from disturbance and sound associated with vessels, to support recovery of SRKWs. Federal vessel regulations were established in 2011 to prohibit vessels from approaching SRKWs within 200 yards and from parking in the path of SRKWs within 400 yards. These regulations apply to all vessels in inland waters of Washington State with exemptions to maintain safe navigation and for government vessels in the course of official duties, ships in the shipping lanes, research vessels under permit, and vessels lawfully engaged in commercial or treaty Indian fishing that are actively setting, retrieving, or closely tending fishing gear (76 FR 20870, April, 14, 2011).

In the final rule, NMFS committed to reviewing the vessel regulations to evaluate effectiveness, and also to study the impact of the regulations on the viability of the local whale watch industry. In December 2017, NMFS completed a technical memorandum evaluating the effectiveness of regulations adopted in 2011 to help protect endangered SRKWs from the impacts of vessel traffic and noise (Ferrara *et al.* 2017). In the assessment, Ferrara *et al.* (2017) used five measures: education and outreach efforts, enforcement, vessel compliance, biological effectiveness, and economic impacts. For each measure, the trends and observations in the five years leading up to the regulations (2006-2010) were compared to the trends and observations in the five years following the regulations (2011-2015). The memo finds that some indicators suggested the regulations have benefited SRKWs by reducing impacts without causing economic harm to the commercial whale-watching industry or local communities, whereas some indicators suggested that vessel impacts continue and that some risks may have increased. The authors also find room for improvement in terms of increasing awareness and enforcement of the regulations, which would help improve compliance and further reduce biological impacts to the whales.

2.3.5 Oil Spills

Because PFMC activities have little to no bearing on the risk of large oil spills due to the PFMC salmon fisheries, we only briefly describe this threat (see NMFS 2008 for more information). In the Northwest, SRKWs are the most vulnerable marine mammal population to the risks imposed by an oil spill due to their small population size, strong site fidelity to areas with high oil spill risk, large group size, late reproductive maturity, low reproductive rate, and specialized diet, among other attributes (Jarvela Rosenberger *et al.* 2017). Oil spills have occurred in the range of SRKWs in the past, and there is potential for spills in the future. Oil can be discharged into the marine environment in any number of ways, including shipping accidents, refineries and associated production facilities, and pipelines. Despite many improvements in spill prevention since the late 1980s, much of the region inhabited by SRKWs remains at risk from serious spills because of the heavy volume of shipping traffic and proximity to petroleum refining centers.

Repeated ingestion of petroleum hydrocarbons by killer whales likely causes adverse effects; however, long-term consequences are poorly understood. In marine mammals, acute exposure to petroleum products can cause changes in behavior and reduced activity, inflammation of the mucous membranes, lung congestion and disease, pneumonia, liver disorders, neurological damage, adrenal toxicity, reduced reproductive rates, and changes in immune function (Schwacke *et al.* 2013; Venn-Watson *et al.* 2015; de Guise *et al.* 2017; Kellar *et al.* 2017), potentially death and long-term effects on population viability (Matkin *et al.* 2008; Ziccardi *et al.* 2015). For example, 122 cetaceans stranded or were reported dead within 5 months following the Deepwater Horizon spill in the Gulf of Mexico (Ziccardi *et al.* 2015). An additional 785 cetaceans were found stranded from November 2010 to June 2013, which was declared an Unusual Mortality Event (Ziccardi *et al.* 2015). In addition, oil spills have the potential to adversely impact habitat and prey populations, and, therefore, may adversely affect SRKWs by reducing food availability.

3 SRKWS AND CHINOOK SALMON FISHERIES

Here we provide a basic description of the relationship between SRKWs and Chinook salmon and a summary of the history of fisheries impacts analyses on SRKWs.

3.1 Relationship between SRKWs and Chinook salmon

As summarized in Section 2.3.1, Chinook salmon have been identified as the SRKWs' primary prey. Several studies in the past have found correlations between Chinook salmon abundance indices and SRKW demographic rates at a coarse coastwide scale (Ford *et al.* 2005; Ford *et al.* 2009; Ward *et al.* 2009; Ward *et al.* 2013). Although these studies examined different demographic responses related to different Chinook salmon aggregate abundance indices, they all found significant positive relationships (high Chinook salmon abundance coupled with high SRKW fecundity or survival). However, these correlations have weakened with the addition of data from recent years. There are several challenges to quantitatively characterize the relationship between SRKWs and Chinook salmon and uncertainty remains. The results of statistical models relating indices of Chinook salmon abundance to measures of SRKW demographic rates are sensitive to which animals and which years are used (*e.g.* only data after 1976 versus only data after 1980), whether Chinook salmon is included as a covariate on specific SRKW demographic metrics like survival or fecundity (and which lag time is used), or the specific Chinook abundance indices used in a given analysis (*e.g.* the Pacific Salmon commission's Chinook Technical Committee's dataset, the Council's FRAM model, etc.) used. Attempts to date to compare the relative importance of any specific Chinook salmon stocks or stock groups using the strengths of statistical relationships have not produced clear distinctions as to which are most influential, and most Chinook salmon abundance indices are highly correlated with each other. Different Chinook salmon populations are more important in different years and the relative importance of specific Chinook salmon stocks in the SRKWs' diet changes over time. If anything, large aggregations of modeled Chinook salmon stocks that reflect abundance on a coastwide scale appear to be equally or better correlated with SRKW vital rates than smaller aggregations of Chinook salmon stocks, or specific stocks such as Chinook salmon originating from the Fraser River that have been positively identified in diet samples as key sources of prey for SRKWs during certain times of the year in specific areas (see Hilborn *et al.* 2012; Ward *et al.* 2013).

There are numerous challenges to identifying statistically robust relationships in natural systems. Demographic stochasticity can create year-to-year variation in measured SRKW demographic rates that mask underlying probabilities or rates. Effects of demographic stochasticity are particularly pronounced because SRKWs have a small population size (*e.g.*, not many births or deaths per year to correlate with annual salmon abundance). These whales are long-lived, thus changes in mortality rates across years are relatively small, making it more challenging to detect statistically-significant changes in mortality rates. Demographic performance across years is also affected by changes in other primary threats (disturbance from vessels and sound and high levels of toxic pollutants) and these effects can confound analysis of the effects of prey abundance. There are substantial uncertainties in the annual Chinook salmon abundance estimates being used to predict SRKW performance, and there is currently no widely-accepted single metric for prey abundance and accessibility to the whales. These challenges make it more difficult to accurately predict the relationship between SRKW demographic rates and Chinook salmon abundance. Nonlinear or threshold responses may not be captured well by relatively simple statistical models.

3.2 History of salmon fisheries impacts analyses

3.2.1 Summary description of the 2009 NMFS biological opinion

In the 2009 biological opinion on PFMC fisheries (NMFS 2009), NMFS compared prey potentially available to SRKWs with and without the action and found that the fisheries will reduce prey available in some locations during some time periods. The analysis considered whether effects of that prey reduction may reduce the reproduction, numbers, or distribution of SRKWs, pursuant to NMFS' jeopardy standard. NMFS evaluated the potential effects of the FMP on SRKWs based on the reductions in prey resulting from a range of harvest scenarios that have been previously authorized, and considered likely in the future, under the FMP.

NMFS evaluated the potential short-term or annual effects as well as the long-term effects of prey reduction from the FMP. Short-term or annual effects of the FMP on prey availability were evaluated as: 1) the percent reduction in Chinook salmon available with the action, and 2) the remaining prey base of Chinook salmon with the action compared to the metabolic needs of the SRKWs. NMFS evaluated the potential for long-term effects on prey availability based on NMFS' most recent conclusions for effects of the FMP on salmon and review of conservation objectives for individual Chinook salmon stock groups affected by the action. The prey reduction was evaluated by time and area, among other factors, based on the information available to stratify the analysis.

Information on Chinook salmon availability was based on FRAM runs. FRAM provides year-specific ocean abundance estimates based on fishery data, escapement estimates, and assumptions about incidental and natural mortality from central California to Southeast Alaska. All Chinook stocks modeled in FRAM travel through the range of SRKWs. FRAM includes most listed and non-listed Chinook stocks within the whales' range, with notable exceptions including Alaska stocks, Upper Columbia River spring, Snake River summer/spring, Klamath River Chinook, Rogue River Chinook, San Joaquin fall, Central Valley late-fall, winter, and spring runs, and fish from other rivers along the Southern Oregon and Northern California coasts. FRAM is a single-pool model that does not provide abundance estimates of Chinook within sub-regions. However, by using catch distribution patterns from the FRAM base period (for the 2009 biological opinion the base period was 1979-1982) when fisheries were broadly distributed across time and area, NMFS developed a method to estimate abundance for inland waters (Strait of Juan de Fuca, east to Georgia Strait in the north, and Puget Sound in the south), and coastal waters (all FRAM fishery regions except inland waters).

Regional abundance estimates were derived for two retrospective years that represented a range of high (2002) and low (2008) Chinook abundance and respective harvest levels. For both years, the estimates were specific to time periods in the FRAM for an annual cycle: October to April, May to June, and July to September. The range of high and low years analyzed was expected to represent a reasonable range of abundance and harvest under the FMP in future years. In general, the percent reduction in Chinook abundance from fisheries is greater in high Chinook abundance years than in low abundance years, because more fish can be caught in high abundance years while still meeting management objectives.

The PFMC salmon fisheries were found to cause minimal or no prey reduction during the October to April time period, regardless of year or region and causes incrementally larger prey reductions during May to June and July to September when the majority of FMP fisheries occur. NMFS' opinions on effects of FMP fisheries on salmon also consider the effects of environmental variability on sustainability of salmon stocks (*i.e.*, from ocean conditions or climate effects) and aim to maintain stocks at or above conservation objectives. Although in specific cases, for some years and stocks the conservation objectives are not met, overall NMFS determined that effects to the ESU still meet ESA compliance standards. When necessary to ensure that the FMP fisheries do not exceed ESA jeopardy standards, regulations for those fisheries have been adjusted to incorporate conservation measures. For example, in 2008 and 2009, poor performance of Chinook salmon stocks in Central Valley, California were the impetus behind fisheries closures south of Cape Falcon, Oregon. As a result of the fishery closures the proposed action would not affect escapements of these stocks. However, while the salmon harvest is managed to meet objectives to promote recovery of salmon, NMFS was not able to evaluate if recovery levels identified for salmon ESUs are consistent with the prey needs and recovery objectives for SRKWs.

NMFS concluded in the 2009 biological opinion that the extent of take was not anticipated to appreciably reduce the survival and recovery of SRKWs. The amount of anticipated take would not increase the risk of mortality (*i.e.*, and therefore would not rise to the level of serious injury or mortality), or hinder the reproductive success of any individual SRKW (NMFS 2009).

3.2.2 Summary description of the 2012 Independent Science Panel review

Following the 2010 Puget Sound Chinook harvest Biological Opinion (NMFS 2011), an independent Science Panel (Panel) reviewed the best available scientific information on the effects that salmon fisheries may have on SRKWs by reducing their prey (Hilborn *et al.* 2012). The Panel and workshop participants reviewed the ecology of the SRKWs, their feeding preferences, and their energy requirements. The participants examined the extent to which various salmon fisheries may reduce prey available to SRKWs, and the potential consequences to their survival and recovery. Following the independent science panel approach on the effects of salmon fisheries on SRKWs, NMFS and partners have actively engaged in research and analyses to fill gaps and reduce uncertainties raised by the Panel in their report.

For reference, below are the key points and conclusions from the Panel report (Hilborn *et al.* 2012), The Workgroup has included some updates based on scientific information that have become available since the Panel report.

Status of Southern Resident Killer Whales

Panel Key Point: The SRKW population has been observed to increase at an average rate of 0.71 percent per year, and would be expected to increase at about one percent per year in the long term if sex ratio at birth were 50:50.

Panel Key Point: The Panel believed that the existing delisting criterion of 2.3 percent growth rate is unlikely to be achieved given current (2012) circumstances or by reducing Chinook salmon fisheries. But if the total abundance continued to increase, a point will be reached where a reappraisal of their status would be likely.

The Panel examined the then-current knowledge of the SRKW population size, growth rates, and demography to: 1) assess current trends relative to historical trends in abundance; and 2) to evaluate the understanding of the current status of the population relative to recovery goals. The Panel examined the time period from 1974 to 2011 and found the population experienced a realized growth rate of 0.71 percent, from 67 individuals to 87 individuals. However, since 2011, the population has declined to 73 individuals and updated status information and population projections are summarized in the December 2016 ESA 5-year status review (NMFS 2016). As described in the Status of the Species and illustrated in Figure 3.2.a, the population is now expected to decline over the next 50 years. However, we note there is increasing uncertainty as the projection extends beyond the first 10 years and with the small population size and number of births the model output can change substantially with the birth of a small number of calves, particularly female calves.

During the workshop, the Vélez-Espino *et al.* (2014) demographic analysis was preliminary and had not yet been published. More recently, Vélez-Espino *et al.* (2014) used data from 1987 to 2011 and estimated expected SRKW population growth rates at a 0.91 percent annual decline for SRKWs (Figure 3.2.a). Furthermore, the estimated SRKW population size was predicted to decline to 75 individuals in a generation (which is considered 25 years), with an extinction risk of 49 percent and an expected minimum abundance of 15 during a 100-year period. The largest contributor to the variance in population growth rate was the survival of young reproductive females. Also the largest contributor to the uncertainty in population growth was the young reproductive female annual survival. Therefore, Vélez-Espino *et al.* (2014) suggest survival of young reproductive females has the largest influence on population growth and population growth variance.

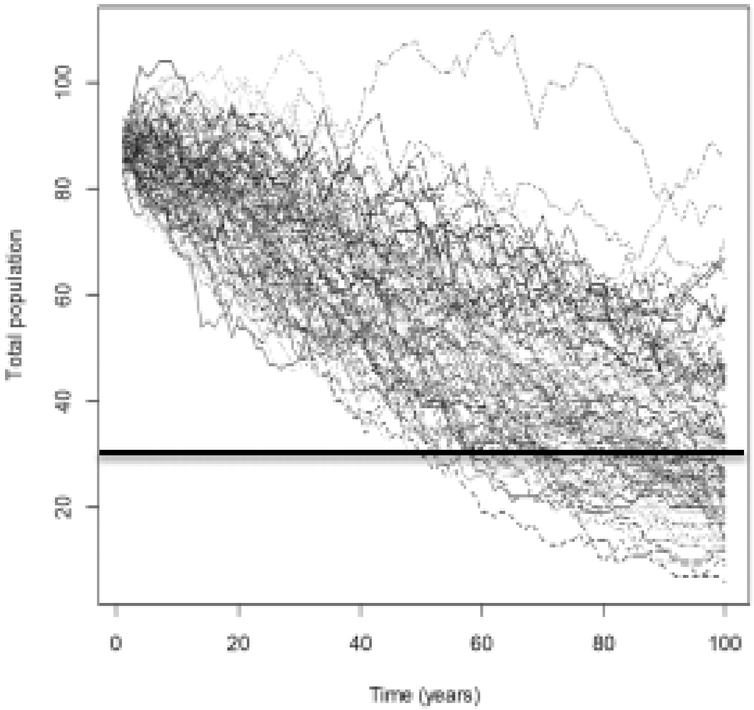

Figure 3.2.a. Projections of SRKW population size under demographic stochasticity and status quo conditions. Horizontal line shows a 30 individual quasi extinction threshold (Vélez-Espino *et al.* 2014).

SRKW Dependency on Chinook Salmon

Panel Key Point: The evidence for strong reliance on Chinook salmon in the summer is convincing, but it is also clear that SRKWs will switch to alternative, more abundant chum salmon when Chinook salmon of suitable size and quality are not readily available in the fall.

Panel Key Point: Photographic evidence supports the assertion that poor condition, which is linked to mortality, and by implication to fecundity, may reflect nutritional stress. However, unless a large fraction of the population experienced poor condition in a particular year, and there was ancillary information suggesting a shortage of prey in that same year, malnutrition remains only one of several possible causes of poor condition.

The Panel report recognized SRKWs have a specialized diet of Chinook salmon from May to September which "means that it is biologically plausible for reduced Chinook salmon abundance to cause nutritional stress and impede recovery of the SRKW population." The report provides context with information on SRKW distribution, diet (species and size selectivity), daily prey requirements, and nutritional stress (Hilborn *et al.* 2012). Despite logistical challenges, the Panel concluded that the diet data collected provide a reasonable indication of what SRKWs are eating in the summer in inland waters; however, winter diet was a major uncertainty. They concluded that Chinook salmon appears to dominate SRKW summer diet and diet is skewed in general towards larger Chinook salmon (4 and 5 year olds). Prey sampling relies on collecting prey remains at or near the surface following a predation event; however, smaller Chinook salmon may not be readily shared with other individuals at the surface and thus could bias detecting their presence. Also fish swallowed at depth could go undetected at the surface and thus not be observed or collected. As discussed in the Status of the Species section above, Ford *et al.* (2016) used fecal DNA analysis to confirm the results of previous studies conducted using other prey identification methods. These fecal samples are thought to be less biased than prey samples recovered from foraging events at the surface because the samples would include information about prey consumed throughout the water column and may also provide information on multiple feeding events.

The Panel considered the bioenergetic modeling approach (Noren 2011) and believed it is a reasonable way to estimate the energy needs of the whales. In contrast, forage ratios (the whales' bioenergetics needs compared to prey available) provide little insight into prey limitations and would require knowing the whale fitness/vital rates as a function of the supply and demand in order for the ratios to be useful. The Panel summarized that of 13 members of the population documented to be in poor condition at that time, all but two died, suggesting some SRKW have been nutritionally limited at certain times of the year. They suggested changes in social behavior may be a sensitive indicator of nutritional limitation.

Fisheries and Prey Availability

Panel Key Point: The maximum long-term increases in abundance of Chinook salmon that might theoretically be available to SRKW would be achieved by eliminating all ocean fishing (typically at least 20 percent increase in ocean abundance of age-4 and age-5 hatchery and wild fish due to elimination of ocean fishery interception of immature fish) and by maximizing recruitment through manipulation of freshwater exploitation rates to maximize recruitment (6 – 9 percent increase in recruitments of wild fish; no impact on hatchery fish).

The best potential for increased Chinook salmon abundance is restoration of freshwater habitat, reducing downstream migration mortality and a change in ocean conditions.

Panel Key Point: The panel sees many potential reasons why not all foregone Chinook salmon catch would be available to SRKW, and is therefore skeptical that reduced Chinook salmon harvesting would have a large impact on the abundance of Chinook salmon available to SRKW.

Projected Future Status and Recovery

Panel Key Point: The statistical analysis by NMFS and DFO scientists are excellent, but the Panel believed considerable caution is warranted in interpreting the correlative results as confirming a linear causal relationship between Chinook salmon abundance and SRKW vital rates.

The Panel described a big picture of the historical vs. current abundances and marine distributions of Chinook salmon; recent trends in Chinook abundance and fisheries; and a description of the probable overlap of SRKW distribution with the distribution of salmon stocks. The Panel considered the results from the correlative approaches that linked Chinook salmon abundance and SRKW vital rates to be consistent with expected dynamics between a predator and its primary prey. The Panel response varied when asked about the strength of evidence that changes in Chinook salmon abundance cause or do not cause changes in SRKW vital rates from being in favor of a cause/effect relationship, rejecting except for one Chinook abundance index, or were unconvinced. The Panel suggested that the regression analyses conducted at the time seemed consistent with a conclusion that SRKW vital rates are more highly correlated with broad scale aggregated abundances of Chinook salmon that overlap with SRKW distribution in spring and late fall periods and potentially winter. However, they concluded a positive relationship between indices of Chinook salmon abundance and killer whale vital rates are probably more complicated than the simple linear relationships assumed. Given the regression results, and the likely higher density of salmon in the inland waters compared to coastal waters, the panel suggested the Chinook salmon that pass through the Salish Sea during the summer period do not directly limit the population growth. Instead, the panel suggested that coastal abundance of Chinook during non-summer months is probably more important for survival and reproduction.

Estimating the Impact of Reducing Chinook Salmon Fisheries on SRKW

Panel Key Point: The Panel was not confident that understanding of the interaction between Chinook salmon fisheries, other predators and SRKW vital rates, is sufficient to expect the model predictions of increased SRKWs to be accurate. The Panel expects the model predictions to overestimate the impact of reductions in Chinook salmon catch on SRKW.

The Panel agreed the methods presented at the workshop seem appropriate for assessing short-term impacts reduced fishing might have on ocean and terminal abundances of Chinook salmon stocks. Using the Fisheries Regulation Assessment Model (FRAM), if ocean fishery exploitation rates were reduced to zero, there would be an expected increase in abundance (both ocean and terminal) of 18 – 25 percent. They emphasized this was assuming no competing risks of death[4],

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[4] The Panel had concerns how natural mortality (and predation on Chinook salmon by SRKW and NRKW) in the FRAM model structure was treated and suggested that a 'competing risks of death' framework that modeled the

implying that this would not be the actual percent increase in abundance of Chinook salmon due to other mortality, such as predation by other species. The Panel noted a 20 percent increase from cessation of all ocean fishing is likely the upper limit of abundance increase and that when Chinook salmon are at lower abundance levels or competing predators are at higher abundance levels, this percent increase would be smaller.

When asked what is the strength of evidence that changes in fisheries in the future would cause or would not cause changes in Chinook salmon abundance sufficient to affect SRKW vital rates, a couple of panelists suggested that any causal effect would be weak, another suggested that changes in fisheries harvest should only be considered for those salmon stocks for which a causal relationship has not been rejected. Lastly several Panel members suggested the impacts on SRKWs from changes to Chinook salmon fisheries would need to consider how this might increase availability of salmon to other predators (*e.g.* NRKWs and pinnipeds).

The Conclusions of the Panel

The Panel believed that the estimated benefits of reducing Chinook salmon harvest in NMFS's analyses provided a maximum estimate of the benefits to SRKWs — and that the realized benefits would likely be lower and insufficient to increase SRKW growth rates to a level that meets existing delisting criteria in the foreseeable future. The Panel concluded that there is good evidence that Chinook salmon are a very important part of the diet of SRKWs and that there is good evidence, collected since 1994, that some SRKWs have been in poor condition and poor condition is associated with higher mortality rates. There is a statistical correlation between SRKW survival rates and some indices of Chinook salmon abundance. Based on those correlations, increases in Chinook salmon abundance would lead to higher survival rates, and therefore higher population growth rates of SRKWs. However, the effect is not linear as improvements in SRKW survival would be expected to diminish at Chinook salmon abundance levels beyond the historical average. Using the statistical correlations, consistently positive SRKW growth rates can occur by avoiding extremely low Chinook salmon abundance levels observed in the 1970-80s and late-1990s.

Elimination of all ocean fisheries for Chinook salmon would impact Chinook salmon abundance far less than the inter-annual Chinook salmon abundance variations that have been seen since the 1970s. The Panel cautioned against overreliance on the correlative studies, and noted that the level of correlation is highly dependent on the choice of Chinook salmon abundance indicators, concluding that the impact of reduced Chinook salmon harvest on future availability of Chinook salmon to SRKWs is not clear.

3.2.3 Summary description of the 2019 NMFS pre-season assessment of fisheries impacts on SRKW

NMFS reinitiated consultation on the 2009 opinion in April 2019. Pending completion of the reinitiated consultation and before adoption of final management measures for 2019, NMFS assessed the impact of 2019 PFMC salmon fisheries on SRKWs. NMFS considered all the information currently available to assess these impacts including:

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effects of fisheries and competing marine mammals on potential consumption of Chinook salmon by killer whales would be more informative.

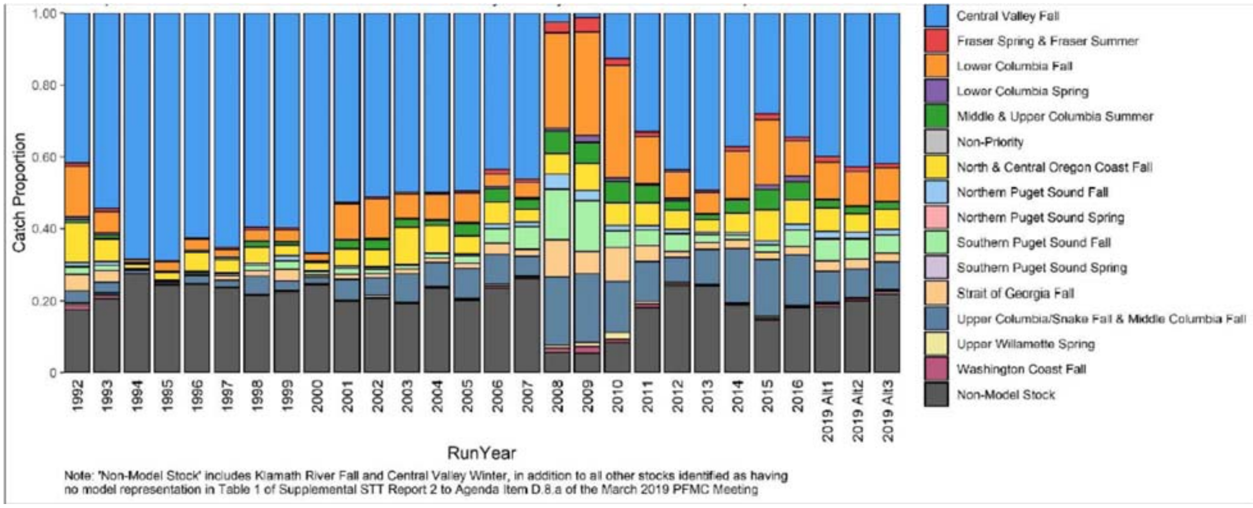
- Estimated percent reductions in overall Chinook salmon prey availability from the March 2019 Council's three fishery alternatives compared to past percent reductions;
- Estimates of 2019 Chinook salmon abundance in coastal waters and inland waters derived using the Chinook FRAM as well as forecasts of Klamath River Fall Chinook and Sacramento River Fall Chinook;
- Supplemental Salmon Technical Team Report 2;
- 2019 pre-season translated forecasts of abundance for each priority Chinook salmon prey stock that contributes to the Council salmon fisheries; and the contribution rates of the priority Chinook salmon prey stocks to total catch (both current predicted contribution and historical contribution) in the Council salmon fisheries.

For 2019, NMFS assessed the effects of the percent reductions to available Chinook salmon prey expected to result from the three fishery alternatives at the March Council meeting under consideration and considered this together with pre-season Chinook salmon abundance estimates for 2019 using FRAM and the two California stock-specific indices mentioned above (Agenda Item F.1.e, Supplemental NMFS Report 1, April 2019). To put the reductions in context, the analysis involved comparing percent reductions in Chinook salmon prey availability from the fisheries and Chinook salmon abundance anticipated in 2019 to percent reductions and abundance for a retrospective time period (NMFS used 1992-2016 as the retrospective time period).

Overall, total percent reductions in prey availability in coastal waters anticipated from each fishing alternative considered by the Council for 2019 ranged from 7.1 percent in Alternative 3 to 9.9 percent in Alternative 1, which fall within the middle range (the range between the lower and upper quartile boundaries) of what was observed during the retrospective time period (1992 – 2016).

Pre-season coastal Chinook salmon abundance and inland Chinook salmon abundance were estimated to fall within a middle range of abundances estimated during the retrospective time period. Therefore, coastal and inland Chinook salmon abundances projected for 2019 were not in the low nor high quartiles for abundances compared to previous years. NMFS also assessed the forecasted pre-season abundances of the priority Chinook salmon prey stocks relative to past abundances during the same retrospective time period (1992 to 2016). Four priority stocks were anticipated to have relatively high Chinook salmon abundances (above the upper quartile boundaries) and ten stocks were anticipated to be within a middle range of abundances (*i.e.*, neither substantially low nor high). Therefore, 2019 abundance estimates for 14 of the 16 priority prey stocks contributing to Council-area salmon fisheries were expected to be in the middle or upper quartiles of abundance when compared with the retrospective time period. Two priority Chinook salmon prey stocks, the lower Columbia River spring and the upper Willamette spring, have abundance estimates in the lowest quartile compared to the retrospective time period.

NMFS focused on these two priority stocks to help assess if the impacts of the 2019 Council area fisheries on these stocks would result in a level it deemed as unacceptable risk by increasing mortality or reducing fecundity of SRKWs because of the stocks' relatively low 2019 abundance compared to their abundances over the retrospective time period. The lower Columbia River spring stock is a low abundance stock but considered high priority because of its spatial and temporal overlap with the whales and because it has been observed in the whales' diet during the winter period when the whales may have a higher likelihood of reduced body condition. However, the stock is a minor contributor to the catch composition of Council area salmon fisheries. Over the retrospective time period, this stock contributed to approximately 0.5 percent of the annual catch on average in Council Area fisheries (Figure 3.2.b). Of note, Figure 3.2.b reflects proportional catches in fisheries as they occurred in a given year, and as a result it includes effects of changes in fisheries management as they may have occurred. For example, in 2009-2010, PFMC fisheries in areas South of Cape Falcon were either highly constrained or closed; as a result of that, the proportion of Central Valley and other more southerly stocks in the overall PFMC catch was very low, and proportions of stocks occurring in fishery areas that remained open were higher. In 2019, the percent contribution to the annual catch of the lower Columbia River spring Chinook stock under each alternative is estimated as 0.1 percent (Figure 3.2.b).



Note: 'Non-Model Stock' includes Klamath River Fall and Central Valley Winter, in addition to all other stocks identified as having no model representation in Table 1 of Supplemental STT Report 2 to Agenda Item D.8.a of the March 2019 PFMC Meeting

Figure 3.2.b. Composition of total Council Area Chinook salmon catch by Priority Chinook salmon stock group (PFMC Agenda Item F.1.e Supplemental NMFS Presentation 1, April 2019).

The upper Willamette spring Chinook salmon stock has not been observed in the diet of SRKWs, and thus is further down the priority prey list, but the stock has the same overlap in space and time with the whales as the lower Columbia River spring stock. This stock is more abundant than the lower Columbia River spring Chinook salmon stock, but still considered relatively less abundant when compared to other priority Chinook salmon prey stocks, such as Southern Puget Sound fall, Lower Columbia River fall, and Strait of Georgia fall, among others. The expected contribution of the upper Willamette spring Chinook stock to the catch in 2019 is similar to the historical contribution of this stock to the Council salmon fisheries catch, which averaged less than 0.5 percent during the retrospective time period. Thus, although two priority stocks were anticipated to have low abundance relative to previous years, because of their low occurrence in Council fisheries, NMFS did not anticipate the Council fisheries would substantially reduce the availability of those priority Chinook salmon prey stocks to the whales. Furthermore, the overall forecast composition in 2019 contained a higher proportion of Chinook salmon stocks that are considered to be higher priority than the average composition in the retrospective time period.

4 PFMC SALMON FISHERIES

The Pacific Coast Salmon FMP guides management of salmon fisheries in Federal waters known as the Exclusive Economic Zone (EEZ) 3 to 200 nautical miles off the coast of Washington, Oregon, and California. Salmon of U.S. and Canadian origin are included except in the case of species which are managed in those waters by another management entity with primary jurisdiction (*i.e.,* sockeye and pink salmon by the Fraser River Panel of the Pacific Salmon Commission in the Fraser River Panel Area (U.S.) between 49°N latitude and 48°N latitude). The FMP covers the coastwide aggregate of natural and hatchery salmon encountered in ocean salmon fisheries, but only has management objectives and allocation provisions for Chinook or king salmon (*Oncorhynchus tshawytscha*), coho or silver salmon (*O. kisutch*), and pink salmon(*O. gorbuscha*). Catches of other salmon species are inconsequential (low hundreds of fish or less each year) to very rare (PFMC 2016). In the event this situation should change, management objectives for these species could be developed and incorporated by plan amendment. The incidental harvest of these salmon species can be allowed or restricted under existing federal fishery regulations.

Chinook and coho are the species caught in the greatest numbers in Council-managed ocean salmon fisheries. In odd-numbered years, catches of pink salmon can also be significant, primarily off Washington and Oregon (PFMC 2018a).

The FMP also includes identification of essential fish habitat (EFH) for Chinook, coho, and pink salmon in ocean, estuary, and freshwater, and contains recommendations for measures to avoid or mitigate for impacts to salmon EFH (see PFMC 2016, Appendix A), and a description of the social and economic fishery characteristics (see PFMC 20126, Appendix B).

To the extent practicable, the Council has partitioned the coastwide aggregate of Chinook, coho, and pink salmon into various stock components and complexes with specific conservation objectives. A detailed listing of the individual stocks and stock complexes managed under the plan is provided in Tables 1-1, 1-2, and 1-3 (PMFC 2016). Stocks designated as hatchery stocks rely on artificial production exclusively, while those designated as natural stocks have at least some component of the stock that relies on natural production, although hatchery production and naturally spawning hatchery fish may contribute to abundance and spawning escapement estimates.

The FMP also contains allocation provisions to regulate how salmon resources are shared among user groups and regions. The FMP management framework allows fishing seasons to be set and managed in a fair and efficient manner. The Council's primary means of meeting the requirements of the Magnuson Stevens Act (MSA) to achieve the optimum yield (OY) from the salmon fishery, meaning the amount of fish that will provide the greatest overall benefit to the Nation, is through maximum sustained yield (MSY), which is defined as the largest long-term average catch or yield that can be taken from a stock or stock complex under prevailing ecological and environmental conditions and fishery technological characteristics (*e.g.*, gear selectivity), and distribution of catch among fleets (50 CFR 600.310). The OY to be achieved for species covered by the FMP is the total salmon catch and mortality (expressed in numbers of fish) resulting from fisheries within the EEZ adjacent to the States of Washington, Oregon, and California, and in the waters of those states (including inland waters), and Idaho, that, to the greatest practical extent within pertinent legal constraints, fulfill the plan's conservation and harvest objectives.

Annually the Council recommends management measures to NMFS that achieve the stock conservation objectives for each stock or stock complex (see PFMC 2016, Chapter 3), while simultaneously seeking to fulfill, to the extent practicable, the harvest and allocation objectives (see PFMC 2016 Chapter 5) that reflect the Council's social and economic considerations. The level of total allowable harvest, the relative harvest levels in various management areas, and the species and stock composition of OY varies annually, depending on the relative abundance and distribution of the various stocks and contingencies in allocation formulas, while also considering ESA guidance from NMFS for ESA-listed species affected by implementation of the FMP.

The Council's annual Review of Ocean Salmon Fisheries (stock assessment and fishery evaluation; SAFE) document and pre-season reports (*e.g.,* PFMC 2019a, 2019b, 2019c, and 2019d) assess and specify the present and historical range of harvests and harvest related mortalities that represent the OY.

4.1 Harvest Controls

Control rules are the metrics used to manage the harvest of stocks to achieve OY and prevent overfishing (as defined under the MSA). Control rules are derived using biological reference points and are used to specify the allowable harvest of stocks based on their abundance and are intended to meet conservation objectives,.

The MSA provides an exception to the requirement for a FMP to specify ACLs and accountability measures for stocks managed under the Pacific Salmon Treaty (PST), an international agreement in which the U.S. participates, however, it is still necessary to specify MSY reference points for these stocks. Pacific salmon stocks subject to fisheries in both the U.S. and Canada are managed under the provisions of the Pacific Salmon Treaty (PST). Natural stocks managed under the provisions of the PST include: (1) Puget Sound pink salmon stocks, (2) most non-ESA-listed Chinook salmon stocks from the mid-Oregon coast to the US/Canada border, and (3) all non-ESA-listed coho stocks except Willapa Bay natural coho. For these stocks, the PST annually places overall limits on fishery impacts and allocates those impacts between the U.S. and Canada. It allows the U.S. and Canada to manage their own fisheries to achieve domestic conservation and allocation priorities, while remaining within the overall limits determined under the PST. Because of these provisions of the PST, and the exception provided by the MSA, it is unnecessary for the FMP to specify ACLs or associated reference points for these stocks. The PST also includes measures of accountability which take effect if annual limits established under the Treaty are exceeded, and further reduce these limits in response to depressed stock status.

The ESA requires federal agencies whose actions may adversely affect listed salmon stocks to consult with NMFS. Because NMFS implements ocean harvest regulations, it is both the action agency and the consulting agency for actions taken under the FMP. To ensure there is no jeopardy as a result of this federal action, NMFS conducts ESA consultations with respect to the effects of ocean harvest on ESA-listed salmon stocks. When the biological consultation results in a "no jeopardy" opinion, NMFS issues an incidental take statement which authorizes a limited amount of take of listed species that would otherwise be prohibited under the ESA. In cases where a "jeopardy" opinion is reached, NMFS develops reasonable and prudent alternatives to the proposed action which also authorizes a limited amount of take, but requires modifications or mitigating components to the original action (*i.e.,* the reasonable and prudent alternatives).

The constraints on take authorized under incidental take statements and reasonable and prudent alternatives are collectively referred to as consultation standards in the FMP. These constraints take a variety of forms including FMP conservation objectives, limits on the time and area during which fisheries may be open, ceilings on fishery impact rates, and reductions from base period impact rates. NMFS may periodically revise consultation standards and NMFS annually supplies a guidance letter to PFMC which reflects the most current information.

Because of the need to meet all FMP control rules and ESA consultation standards in each fishing year, Council salmon fisheries are managed under a "weak stock" approach. In order to meet all control rules and consultation standards for the weakest stocks in a given year, Council fisheries forego full use of available harvests for healthier stocks. As a result, it is a very common case for stock-specific harvests for some stocks to be less than allowed under FMP control rules or ESA consultation standards due to the need to protect co-occurring limiting stocks.

4.2 Overall Fishery Objectives

The following FMP objectives guide the Council in establishing fisheries against a framework of ecological, social, and economic considerations.

1. Establish ocean exploitation rates for commercial and recreational salmon fisheries that are consistent with requirements for stock conservation objectives and ACLs within Section 3, specified ESA consultation or recovery standards, or Council adopted rebuilding plans.
2. Fulfill obligations to provide for Indian harvest opportunity as provided in treaties with the U.S., as mandated by applicable decisions of the federal courts, and as specified in the October 4, 1993 opinion of the Solicitor, Department of Interior, with regard to federally recognized Indian fishing rights of Klamath River Tribes.
3. Maintain ocean salmon fishing seasons supporting the continuance of established recreational and commercial fisheries while meeting salmon harvest allocation objectives among ocean and inside recreational and commercial fisheries that are fair and equitable, and in which fishing interests shall equitably share the obligations of fulfilling any treaty or other legal requirements for harvest opportunities.[5]
4. Minimize fishery mortalities for those fish not landed from all ocean salmon fisheries as consistent with achieving OY and the bycatch management specifications of Section 3.5.
5. Manage and regulate fisheries so that the OY encompasses the quantity and value of food produced, the recreational value, and the social and economic values of the fisheries.
6. Develop fair and creative approaches to managing fishing effort and evaluate and apply effort management systems as appropriate to achieve these management objectives.
7. Support the enhancement of salmon stock abundance in conjunction with fishing effort management programs to facilitate economically viable and socially acceptable commercial, recreational, and tribal seasons.
8. Achieve long-term coordination with the member states of the Council, Indian tribes with federally recognized fishing rights, Canada, the North Pacific Fishery Management

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[5] In its effort to maintain the continuance of established ocean fisheries, the Council includes consideration of maintaining established fishing communities. In addition, a significant factor in the Council's allocation objectives in Section 5.3 is aimed at preserving the economic viability of local ports and/or specific coastal communities (*e.g.,* recreational port allocations north of Cape Falcon). Chapter 6 in Appendix B and the tables it references provides additional specific information on the fishing communities.

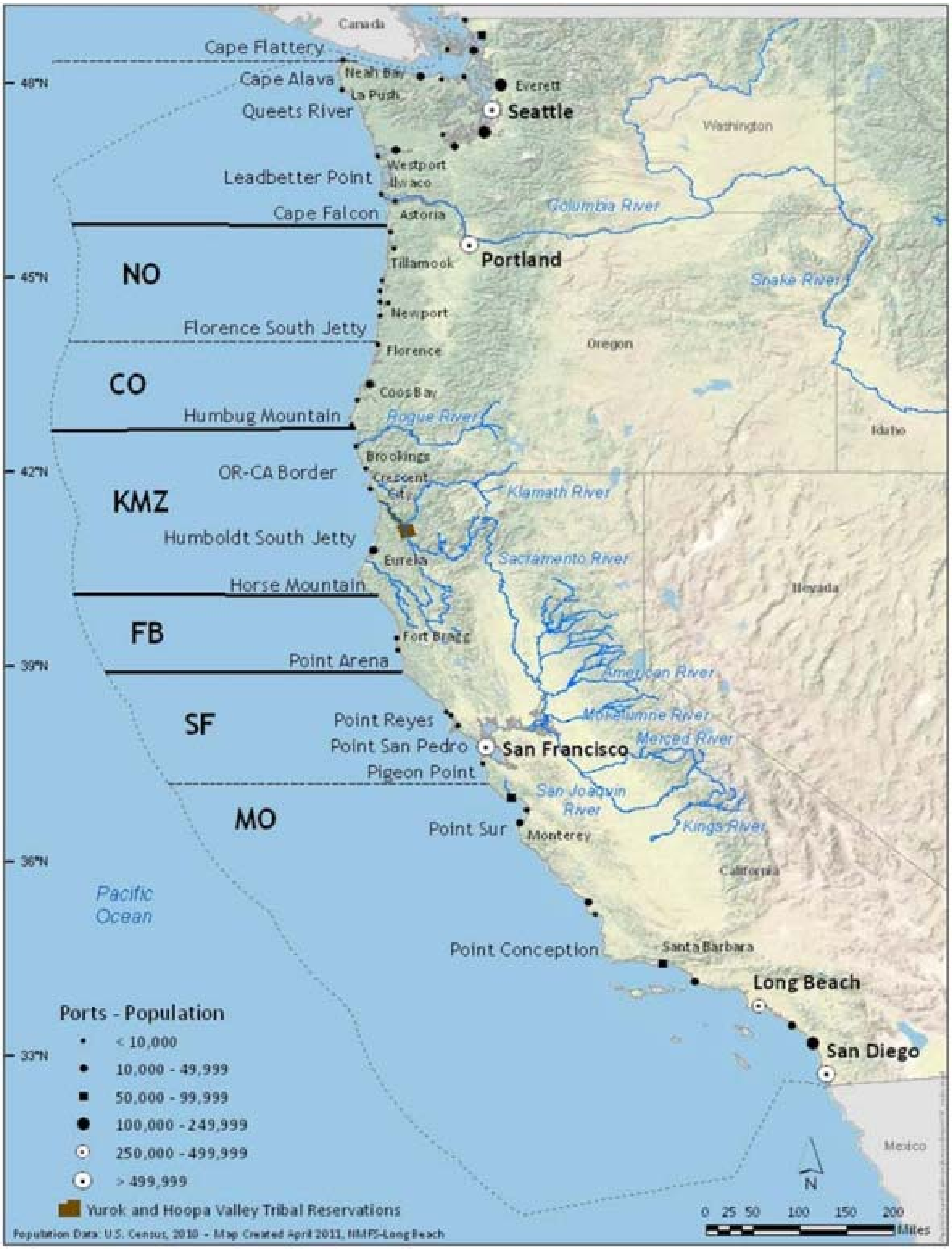
Council, Alaska, and other management entities which are responsible for salmon habitat or production. Manage consistent with the PST and other international treaty obligations.

9. In recommending seasons, to the extent practicable, promote the safety of human life at sea.

Harvest allocations are determined from a total allowable ocean harvest, which is maximized to the largest extent possible but still consistent with PST and treaty-Indian obligations, state fishery needs, and spawning escapement requirements, including consultation standards for stocks listed under the ESA. The Council makes every effort to establish seasons and gear requirements that provide commercial troll and recreational fleets a reasonable opportunity to catch the available harvest. Procedures for determining allowable ocean harvest vary by species, area, fishery complexity, available data, and the state of development of predictive tools. These procedures have and will change over time to incorporate the best available science. A number of management controls are available to manage the ocean fisheries each season, once the allowable ocean harvests and the basis for allocation among user groups have been determined. Stock management considerations also guide the Council for setting seasons within major subareas of the Pacific Coast (Figure 4.2.a).

Controls include management boundaries and seasons, quotas, minimum harvest lengths, fishing gear restrictions, area restrictions, commercial landing limits, and recreational daily bag limits. Natural fluctuations in salmon abundance require that annual fishing periods, quotas, and bag limits be designed for the conditions of each year. Measures that are suitable one year may not be suitable the next. New information on the fisheries and salmon stocks also may require other adjustments to the management measures. The Council assumes these ocean harvest controls also apply to territorial seas or any other areas in state waters specifically designated in the annual regulations. Details to the incorporation and use of these controls are contained in Chapter 6 of the FMP (PFMC 2016).

Successful management of the salmon fisheries requires considerable information on the fish stocks, the amount of effort for each fishery, the harvests by each fishery, the timing of those harvests, and other biological, social, and economic factors. Much of the information must come from the ocean fisheries; other data must come from inside fisheries, hatcheries, dam counts, and spawning grounds. Some of this information needs to be collected and analyzed daily, whereas other types need to be collected and analyzed less frequently, *i.e.*, once a year. In general, the information can be divided into that needed for in-season management and that needed for annual and long-term management. The methods for reporting, collecting, analyzing, and distributing information can be divided similarly. The description of the data needs, methods for obtaining in-season and annual long-term data, reporting requirements, and schedules for the Council's monitoring of the resource and the fisheries harvesting that resource are contained in Chapters 7 and 8 of the FMP (PMFC 2016).



Population Data: U.S. Census, 2010 - Map Created April 2011, NMFS-Long Beach

Figure 4.2.a. Map of major management boundaries in common use since 2000.

4.3 Schedule and Procedures for establishing annual management measures

The process for establishing annual or pre-season management measures under the FMP contains a considerable amount of analysis, public input, and review. This is detailed in Chapter 9 (PFMC 2016). The actions by the Secretary of Commerce after receiving the pre-season regulatory modification recommendations from the Council are limited to accepting or rejecting, in total, the Council's recommendations. If the Secretary rejects such recommendations he or she will so advise the Council as soon as possible of such action along with the basis for rejection, so that the Council can reconsider. Until such time as the Council and the Secretary can agree on modifications to be made for the upcoming season, the previous year's regulations remain in effect. This procedure does not prevent the Secretary from exercising his or her authority under Sections 304(c) or 305(c) of the MSA and issuing emergency regulations, as appropriate, for the upcoming season. In-season modifications of the regulations may be necessary under certain conditions to fulfill the Council's objectives and the process and procedures for doing so are detailed in Chapter 10 (PFMC 2016). Modifications not covered within the framework will require either an FMP amendment, rulemaking, or emergency Secretarial action. Depending on the required environmental analyses, the amendment process generally requires at least a year from the date of the initial development of the draft amendment by the Council. Emergency regulations may be promulgated without an FMP amendment. Details for both an FMP Amendment process and Emergency Regulations are detailed in Chapter 11 (PFMC 2016).

4.4 Season structure

[Pending]

4.5 Chinook salmon harvest

The Annual post-season review of the Council's recommended management measures is documented by the Salmon Technical Team (STT) and Council staff in the stock assessment and fishery evaluation (SAFE) document (for an example refer to PFMC 2019a). The SAFE document reviews the prior year's ocean salmon fisheries off the coasts of Washington, Oregon, and California to assess Council salmon fishery management performance, the status of Council-area salmon stocks, and the socioeconomic impacts of salmon fisheries. The SAFE document compares post-season fishery performance against achieving stock specific conservation objectives and reference points governing harvest control rules and status determination criteria for salmon stocks and stock complexes in the Pacific Coast salmon FMP (for current objectives see Table 3-1, PFMC 2016). Over the course of implementing the FMP many of these objectives have been modified, often as a result of ESA consultations for specific species of salmon. Therefore trying to depict long-term adherence to a single standard or contemporary reference point for many stocks will not capture the changes that may have occurred over several decades, or the changes that have taken place in just the last decade. Similarly, due to the weak stock management approach, evaluation of the effects of FMP control rules and ESA guidance on past fisheries management is a very complex exercise necessitating consideration of the status of all managed stocks in any given year. Nevertheless, an examination of catch and stock status information can be used to describe general trends in PFMC salmon fisheries over time.

As an example, the workgroup examined fisheries effects and Chinook salmon abundance over the period 1992-2016 using the modeling approach described in Section 5.1. The exercise calculated the post-season abundance with and without Council fisheries having occurred, for each calendar year and by geographical area. Figure 4.5.a through Figure 4.5.d show Chinook salmon abundance and fishery mortality removals in Council-area salmon fisheries for the major management areas the Workgroup aggregated: North of Cape Falcon, Oregon coast (Cape Falcon south to Horse Mountain, California) and California coast (south of Horse Mountain).

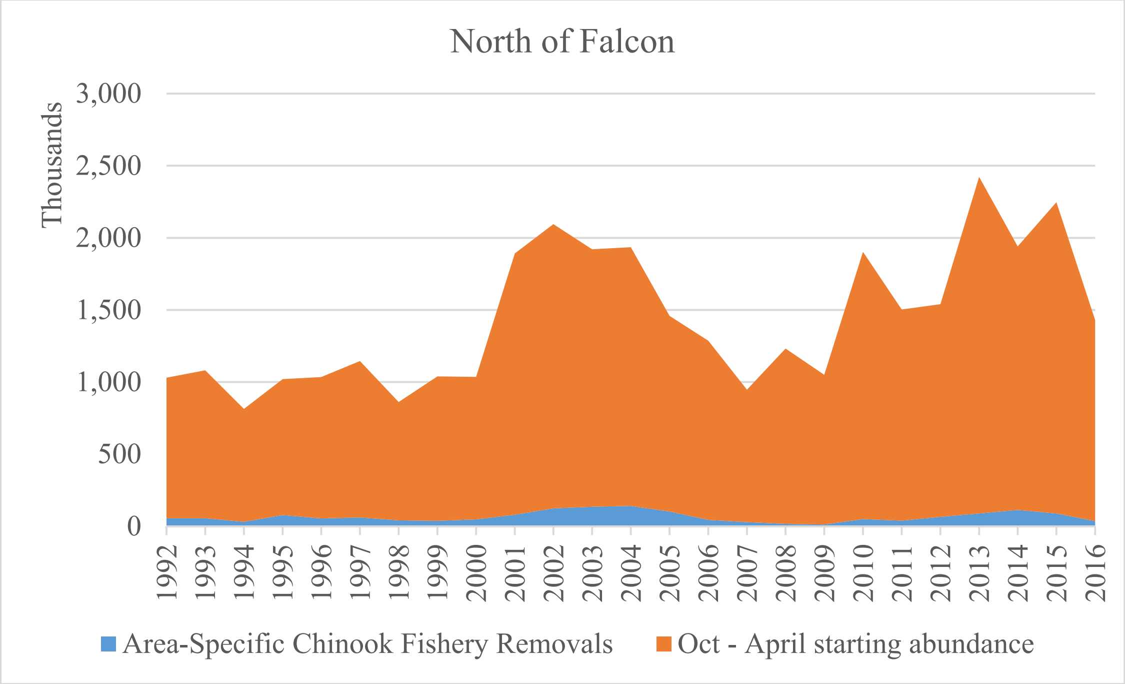


Figure 4.5.a. North of Cape Falcon 1992-2016 trends in annual abundance (estimated annually to be present on October 1) and area-specific fishery adult mortality estimates (from October through the following September).

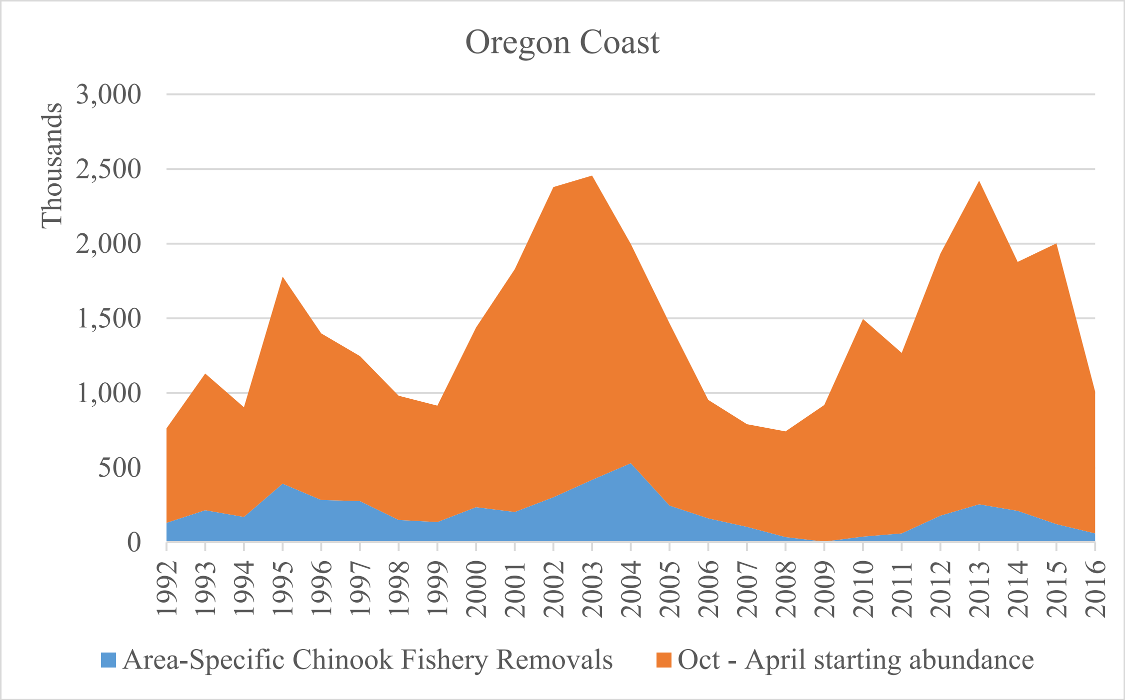


Figure 4.5.b. Oregon coast (Cape Falcon south to Horse Mountain, California) coastal 1992-2016 trends in annual abundance (estimated annually to be present on October 1) and area specific adult fishery mortality estimates (from October through the following September).

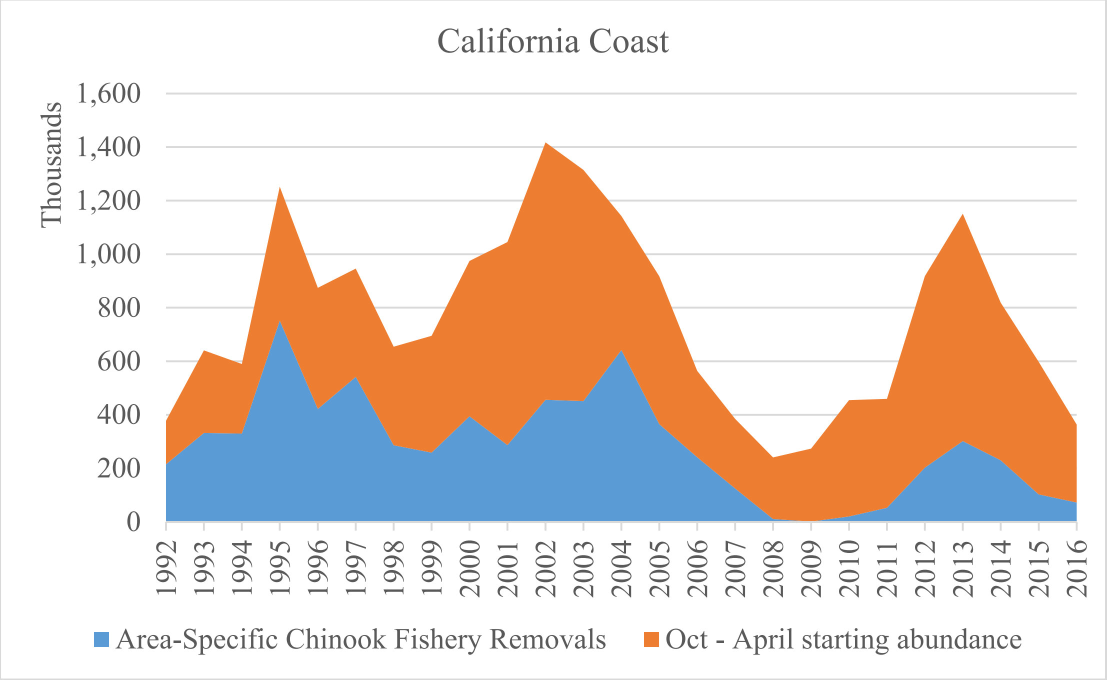


Figure 4.5.c. California coast, south of Horse Mountain coastal 1992-2016 trends in annual abundance (estimated annually to be present on October 1) and area specific fishery adult mortality estimates (from October through the following September).

At a coastwide level Figure 4.5.d. depicts Chinook salmon abundance and fishery mortality removals aggregated across all areas of the EEZ. As described above, this level of abundance has changed over time relative to implementing changes to harvest control rules and ESA limitations on the fisheries. In order to determine a trend, if one exists, you can divide the fishery abundance by the without fishery abundance show the percent reduction in ending abundance that is attributable to PFMC fisheries having occurred. When plotted by year for coastwide abundance, the percent of potential abundance that is remaining after fisheries occur is increasing over time – meaning fisheries have been taking less of the available abundance over time (Figure 4.5.e). The trend line depicted in Figure 4.5.e. is not intended to reflect any particular level of significance, but indicates whether the long term trend is up, down, or flat over time.

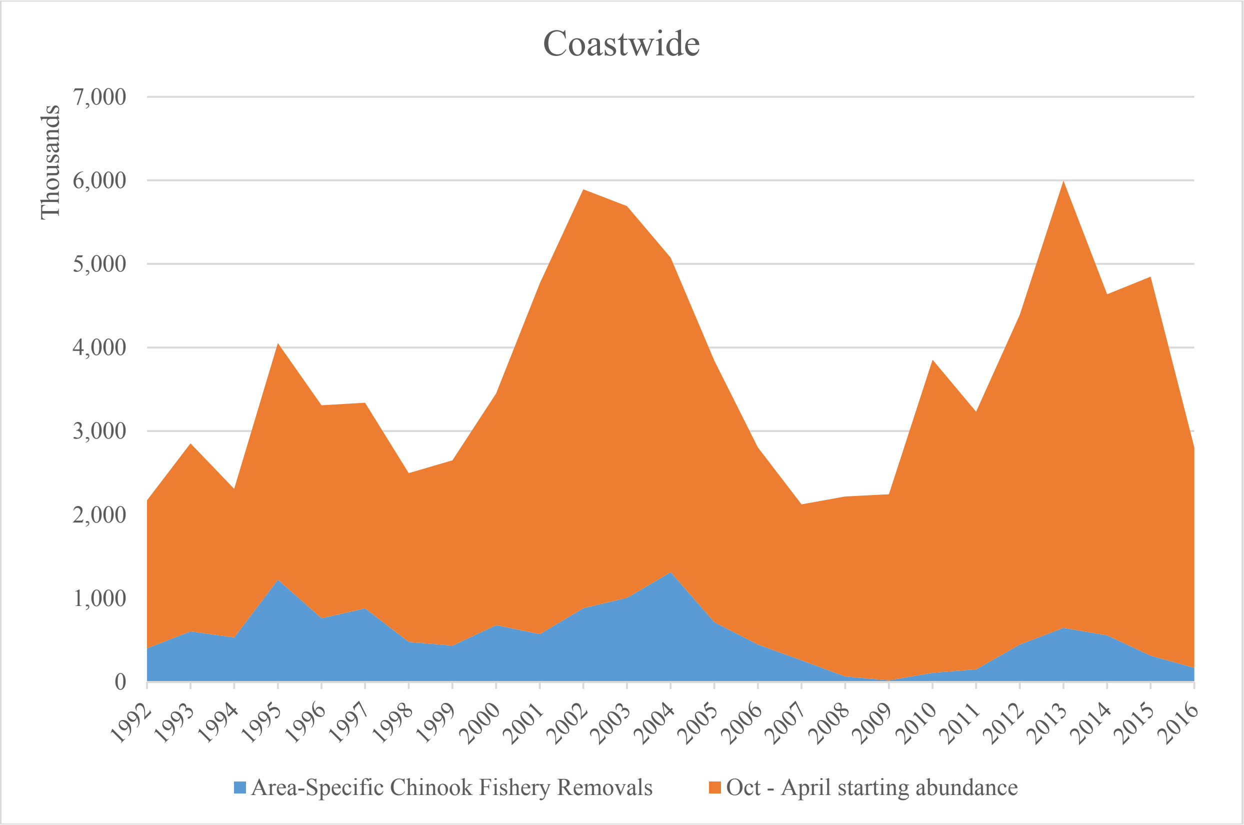


Figure 4.5.d. Coastwide coastal 1992-2016 trends in annual abundance (estimated annually to be present on October 1) and fishery adult mortality estimates (from October through the following September).

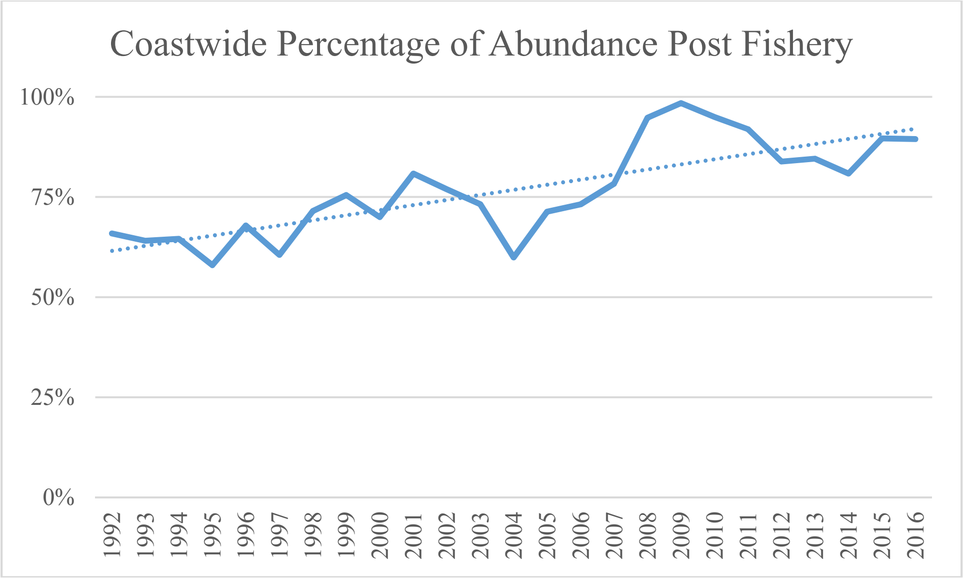


Figure 4.5.e. Coastwide coastal 1992-2016 trend in percent Chinook available after harvest from fishery adult mortality in PMFC salmon fisheries on coastwide coastal salmon abundance (from October through the following September).

Estimates of harvest reductions for each geographic area aggregated for evaluation in this report using the methodology described in Section 5 are contained in Appendix E. They are available annually, also by the time steps described in further detail in Section 5.1**Error! Reference source not found.**.

5 RISK ASSESSMENT

The analyses developed to support the risk assessment use correlative analyses of relationships between Chinook abundance and SRKW demography similar to those included in the Panel Report (Hilborn *et al.* 2012) and described by Ward *et al.* (2013). These new analyses include more recent data and include a broader range of SRKW demographic indices. While previous documents have referred to such relationships as "correlations" we note that strictly speaking we are estimating coefficients in generalized linear models with non-Gaussian distributions rather than calculating correlation coefficients. In contrast with earlier correlative studies, abundances were defined not on the basis of individual stocks or sets of stocks, but on the basis of composite abundances in specific ocean areas based on distributions inferred from recent modeling efforts (Shelton *et al.* 2019). This analysis related past SRKW demographic performance with retrospective estimates of time- and area-specific Chinook salmon abundance. For the first part of the analysis, only the estimated Chinook salmon abundance present in a particular time and area was of interest; no attempt was made to separate out the effects of production, natural mortality, or harvest in generating the realized abundances. We then estimated time- and area-specific fishery removals and used the fitted relationships between Chinook abundance and demographic rates to calculate the change in predicted demographic rates corresponding to fished versus unfished abundances.

As a coarser approach, we also performed clustering analyses attempting to identify sets of years of similar SRKW demographic performance, and then examined whether years of good or bad demographic performance were consistently associated with high or low Chinook abundance. The clustering analyses are described in Appendix D.

5.1 Model Description

The models analyzed the statistical relationship between demographic indices of SRKW performance (see Section 5.2) and retrospective estimates of adult (age 3 and older) ocean Chinook abundance at three time steps (October 1-April 30, May 1-June 30, and July 1-September 30) aggregated at various spatial scales and for fishery management years 1992-2016 (the fishery management year starts in the fall of the preceding year, so the first time step considered was October 1, 1991 – April 30, 1992). When biologically appropriate, we considered temporal lags between Chinook abundance and observed SRKW performance based on plausible physiological mechanisms linking food supply to future performance. For example, because killer whales have a gestation period of approximately 17 to 18 months, it may be important to consider Chinook salmon indices in earlier years as predictors of fecundity (Hilborn *et al.* 2012, Ward *et al.* 2009, Ward *et al.* 2013). While discussed as a potential evaluation consideration, the workgroup decided not to consider moving averages of abundance across multiple years.

Coastwide abundance estimates for most Chinook salmon stocks were generated using Chinook FRAM (MEW 2008) post-season runs (Round 6.2 of base period calibration; 10.29.2018). Abundance estimates for FRAM stocks are calculated using stock-specific terminal run size estimates by age and mark status provided by regional technical staff. Stock-specific terminal run sizes are then expanded by maturation rates, fishing mortality, and natural mortality estimates to derive a starting abundance. For additional details related to calculations of FRAM starting abundances, please refer to the Backwards FRAM documentation, available at https://github.com/dappdrd/PFMC_SRKW/blob/master/BkFRAM-May-2-2018.docx.

However, there are several Chinook stocks that are not modeled in FRAM. These stocks include those north of Vancouver Island, Hupp Springs, Washington Coastal Springs, Tsoo-Yess Falls, Upper Columbia Spring/Snake River spring-summer, and all Chinook salmon stocks originating south of Elk River, Oregon, with the exception of Sacramento River Fall Chinook (SRFC) (see Appendix A and B). Many of these stocks were relatively small in magnitude (*e.g.,* Hupp Springs, Coastal Springs, and Tsoo-Yess) or were primarily outside of the core SRKW assessment area (*e.g.,* stocks north of Vancouver Island). However, the SRKW workgroup determined that it was necessary to account for Sacramento Fall, Klamath Fall, and Rogue Fall stocks along with Upper Columbia Spring/Snake River spring-summer using methods external to FRAM due to the likely spatial-temporal overlap of these stocks with SRKW and relatively large abundances of these stocks.

For Upper Columbia Spring/Snake River spring-summer, terminal run size estimates were expanded to account for assumed ocean natural mortality (using the same natural mortality assumptions as Chinook FRAM) to represent starting abundances. See below:

$$Starting\ Abundance = \frac{Terminal\ Run\ Size}{(1 - natural\ mortality)}$$

Where the natural mortality is age specific to time step 1 ("Oct. through Apr.") and the terminal run size represents the return of Upriver Columbia Springs in time step 1. Starting abundance of Upriver Columbia Springs is only available for time step 1, with starting abundances in time steps 2 and 3 for this stock being set to 0.

The Upriver Columbia Spring aggregate typically experiences an exploitation rate less than one percent in all marine fisheries (the Workgroup estimated this value using a coded wire tag analysis applied to data from 2000–2016). Given the very low rates of ocean exploitation, it is presumed that this stock aggregate either has a far north or offshore distribution, resulting in low encounter rates in fisheries. Therefore, Upper Columbia Spring/Snake River spring-summer are most likely to be available to SRKW near the mouth of the Columbia River as they return to spawn and unavailable outside of the winter season.

For Chinook salmon stocks originating south of the Elk River, we used abundance estimates for SRFC, Klamath River Fall Chinook (KRFC), and Rogue River Fall Chinook (RRFC) derived outside of FRAM. Although SRFC are included in FRAM as Sacramento Falls, we chose to use an alternative model that more closely aligns with South of Falcon fisheries management conventions and models. For SRFC we used a modification of the Sacramento Index (O'Farrell *et al.* 2013) incorporating natural mortality and catch apportioned by month, for KRFC we used the same cohort reconstructions that inform the Klamath Ocean Harvest Model (KOHM; Mohr 2006), and for RRFC we adjusted the September 1 age-specific Rogue Ocean Production Index (ROPI) values (PFMC 2019) according to monthly ratios in age-specific KRFC abundance determined from cohort reconstructions. Additional details are available in Appendix B, and Appendix A discusses stocks for which abundance estimates are not available.

At each time step, coastwide ocean abundances were distributed among spatial boxes based on estimates of the proportion of each stock found in each area each season. For fall run stocks, proportional abundance in each management area was based on the results of Shelton *et al.* (2019). This is a state-space model that infers time- and area-specific ocean abundances of tagged fish from representative coded-wire tagged release groups using information on release size, time- and area-specific fishery catch and effort, and age structure of returning spawners. Individual FRAM stocks were matched up to units of analysis in the Shelton *et al.* model as described in Table 5.1.a. SRFC corresponds with Shelton *et al.*'s SFB stock and KRFC corresponds with NCA. Although the Rogue River is in Southern Oregon, the "SOR" results in Shelton *et al.* are for Chetco River fish. Spatial patterns in recoveries of Rogue River Chinook coded-wire tags (Weitkamp 2010) and genetically-identified fish (Bellinger *et al.* 2015, Satterthwaite *et al.* 2015) are more similar to Klamath River Chinook than to other Southern Oregon Chinook, so we apportioned RRFC spatially using NCA results. For spring run stocks, which lacked distribution estimates from Shelton *et al.*, we followed the logic described in https://www.fisheries.noaa.gov/webdam/download/93036440, using point values of 0.02 to represent ranges of 0-0.05, 0.15 to represent ranges of 0.05-0.25, and 0.5 for areas directly adjacent to the river of origin. Note that, as per the logic above, Upriver Columbia Spring Chinook were considered unavailable to SRKW in all areas during the "May–June" and "July–September" time steps.

Table 5.1.a. Mapping Chinook stocks used within the Shelton *et al.* model to the FRAM model stocks.

Stock (Shelton)	Stocks (FRAM)
Central Oregon	Mid Oregon Coast
Lower Columbia	Columbia River Oregon Hatchery Tule, Columbia River Washington Hatchery Tule, Lower Columbia River Wilds, Lower Columbia Naturals
Middle Columbia	Columbia River Bonneville Pool Hatchery
Upper Columbia	Columbia River Upriver Summer, Columbia River Upriver Bright, and Snake River Fall
Northern Oregon	Oregon North Coast
Puget Sound	Nooksack/Samish, Skagit, Snohomish, Stillaguamish, Tulalip, Mid Puget Sound, University of Washington Accelerated, South Puget Sound, Hood Canal, Juan de Fuca Tributaries, Hoko
Southern Georgia Strait	Fraser Lates, Fraser Earlies, Lower Georgia Strait
Washington Coastal	Willapa Bay, Washington North Coast
West Coast Vancouver Island	West Coast Vancouver Island

We then aggregated individual spatial boxes and their corresponding abundances at three levels:

1. the entire U.S. West Coast EEZ as a single unit ("Coastwide");
2. the West Coast EEZ split into two boxes north versus south of Cape Falcon ("NOF" and "SOF"), and
3. the West Coast EEZ split into three boxes at Cape Falcon and at Horse Mountain, which are among the management area lines used in ocean fisheries management by the PFMC (north of Cape Falcon "NOF", between Cape Falcon and Horse Mountain "Oregon coast", and south of Horse Mountain "California coast"). We

also calculated separate abundances for the Salish Sea (“Salish”; sum of PUSO and SGEO from Shelton *et al.* 2019) and Southwest West Coast Vancouver Island (“SWCVI”). Note that "Coastwide" does not include the Salish Sea nor SWCVI.

5.2 Demographic Indices Considered

The workgroup considered the following demographic indices: 1) SRKW survival rates, 2) SRKW fecundity (birth) rates, and 3) frequency of occurrence of "peanut-head" whales (a metric previously used as an index of extremely poor condition, Matkin *et al.* 2017). The birth and death data were filtered to remove individual-year combinations that were associated with known non-prey related deaths (*e.g.* vessel strike). A number of additional metrics were also discussed, but not ultimately included for a variety of reasons (questionable utility as indicators, few years of data, etc.). The list of these latter metrics included social cohesion (Parsons *et al.* 2009), occupancy of the Salish Sea (Olson *et al.* 2018), changes in body condition other than the occurrence of peanut-head whales (Fearnbach *et al.* 2018), hormone indicators of nutritional status (Wasser *et al.* 2017), indicators based on stable isotope data (Warlick *et al.*, in review), diet diversity (Ford *et al.* 2016), and demographic parameters of Northern Resident killer whales (Olesiuk *et al.* 2005; Ford *et al.* 2009).

SRKW survival varies with age or stage of the whale (Olesiuk *et al.* 1990). Because some ages were uncertain (particularly older animals at the start of the survey), we modeled an effect of stage on survival so that we could compare survival standardized to a common stage across years (Hilborn *et al.* 2012; Ward *et al.* 2013). Similarly, fecundity varies with age so we modeled an effect of age on fecundity so that we could compare fecundity at a common age (set to age 20 because fecundity is thought to peak in the early 20s [Ward *et al.* 2009]).

Similar to previous analyses attempting to link killer whale demography with metrics of Chinook salmon abundance (Ford *et al.* 2009, Hilborn *et al.* 2012), the Workgroup considered temporal lags between Chinook salmon abundance and observed SRKW performance based on plausible physiological mechanisms linking food supply to future performance, and to allow for uncertain timing in the death of SRKW experiencing mortality events. We considered Chinook salmon abundance estimates during the corresponding management year (no time lag) for all three vital rates. We also considered abundance estimates at lags of one year for both survival and fecundity for several reasons including to allow time for plausible biological mechanisms to operate (*i.e.* food stress could lead to reduced body condition and health leading to increased disease susceptibility leading to eventual death).

Additionally, we also considered a lag of one year because exact birth and especially death dates are uncertain, such that births or deaths assigned to a particular calendar may have already taken place before the corresponding management year is complete, and it would not be sensible to relate demographic rates to Chinook salmon abundance measured after the demographic event of interest has already taken place. Because killer whales have a gestation period of approximately 17 to 18 months, it may be important to consider Chinook salmon indices in year t-2 as predictors of fecundity (Hilborn *et al.* 2012, Ward *et al.* 2009, Ward *et al.* 2013) to allow for a lagged response to food supply in the initiation of pregnancy, followed by the extensive gestation period.

It is important to note that we modeled relationships between annual demographic rates and Chinook salmon abundance indices measuring different seasonal time steps within the year, not demographic rates within the specific seasonal time steps. Also, though seasonal metrics of salmon abundance were available, the killer whale response variables have been recorded on an annual basis.

5.3 Model Structure

Fecundity of individual female whales was modeled using logistic regression as a function of time-area specific Chinook salmon abundance along with a quadratic function of age, allowing for fecundity peaking at an intermediate age. Whales that gave birth in the previous year were excluded due to the approximate 17 to 18 month gestation period meaning they could not possibly give birth again the following year (Ward *et al.* 2013). Females younger than 10 or older than 42 were excluded from the fecundity analysis (Ward *et al.* 2013). We separately considered abundance in the current year, in the prior year, and two years prior to account for lagged effects.

Survival of individual whales was modeled using logistic regression as a function of time-area specific Chinook abundance and a categorical variable describing stage/sex (juvenile, young female, young male, old female, old male). For consistency, we used delineations that have been used previously (Ward *et al.* 2013). As discussed in previous workshops (Hilborn *et al.* 2012), to avoid introducing biases, we removed the deaths of a handful of whales whose cause of death was thought to be associated with infection from satellite tags (L95), ship strikes (e.g. J34, L112), or several deaths of juveniles that disappeared with their mothers (and thus thought to not be independent). In all cases, we included survival of these whales up to the year of death, just not the death itself.

The occurrence of whales with peanut-head each year as a function of area-specific Chinook abundance was modeled using Poisson GLM (Poisson family with log-link). Alternative approaches could include logistic regression, for example, but the number of whales with this condition is extremely small such that sample size precludes inclusion of covariates (age, sex) that might explain variation. Thus, all animals were assumed to have equal chances of developing the peanut head syndrome.

All statistical analyses were performed in R (R Core Team 2019). The code and statistical methodology used by the SRKW workgroup to perform all analyses is publicly available and can be accessed at: https://github.com/dappdrd/PFMC_SRKW.

5.4 Model Run Descriptions

Complete results can be obtained from https://dappdrd.shinyapps.io/SRKW_Chinook_Analysis/

To use the application and produce outputs:

1.) Go to it via website: https://dappdrd.shinyapps.io/SRKW_Chinook_Analysis/

2.) Input your email address.

3.) Send the input file to your email via the associated button (this may take a moment).

4.) Save the input file to your computer and then use the browse button on the application to select the input file.

5.) Press the “Begin Processing/Email Outputs” button to send an output file to your email (this may take a few minutes).

Interpreting the results:

1.) Each area-time step can be found as a tab in the output file. Time step 1 corresponds to October-April, time step 2 corresponds to May-June, time step 3 corresponds to July-September.

2.) Each graphic depicts the relationship between Chinook abundance and a SRKW population parameter. Each analysis was conducted as a logistic regression (or Poisson regression in the case of peanut-head). For fecundity analyses, age was included as a covariate and modeled as a quadratic. For the survival analysis, stage was included as a covariate. In order, the analyses are Chinook abundance versus fecundity (no lag; row 2), survival rates (row 28), peanut head (row 54), fecundity (1 year lag; row 80), fecundity (2 year lag; row 106), and survival (1 year lag; row 158).

3.) The model summaries are available to the left of each graphic. To determine if there is a statistically significant relationship between Chinook abundance on each population parameter, refer to the p-value for abundance in these sections.

None of the fitted regressions met the typical criterion of $p \leq 0.05$ that is often associated with "statistical significance" (Table 5.4.a through 5.4.g summarize the regression statistics; Appendix C depicts all regression model outputs). However, several regressions had $p \leq 0.10$, and this occurred for times and areas where whale presence is understood to be most likely (Table 5.4.a). Although $p \leq 0.05$ is the typical criterion for "statistical significance", there is precedent for using other values. Fields such as genomics that generate large amounts of data have a precedent for using smaller values (e.g. Concato and Hartigan 2016), and some statisticians have proposed using values smaller than 0.05 to avoid 'p-hacking' (Benjamin et al. 2018). In the opposite direction, some fields have conventionally used slightly larger values, or interpreted larger p-values as 'marginally significant' (Pritschet et al. 2016). Larger significance thresholds may be considered more appropriate in the face of noisy data, small sample sizes, and/or cases where the consequences of erroneously rejecting an effect are considered more severe than the converse. In addition, mechanistic hypotheses may justify a 1-tailed test (testing the probability under the null hypothesis of a coefficient greater than zero by at least certain amount, rather than the probability of a coefficient at least a certain distance from zero in either direction, resulting in a smaller p value given the same data and model). Similarly, because of multiple statistical tests being conducted with correlated Chinook salmon abundance estimates (abundance in adjoining spatial boxes being generated from the same cohorts and FRAM modeled output) an argument could be made for applying a correction factor (e.g., Bonferroni) that would lead to a higher critical p-value. However, best practices call for making such adjustments before seeing the results, but the workgroup did not do so.

A p-value of 0.05 means that given the level of variability in the data and the model assumptions, there is a five percent probability of seeing a relationship at least as strong as the one observed purely by chance under a null hypothesis of no effect (in this case a parameter value of zero for the coefficient describing the change in demographic SRKW rate with changes in Chinook abundance, again conditional on the assumed form of the statistical model). It should not be interpreted as the probability that there is or is not an effect in any particular case (Wasserstein and Lazar 2016). Rather, a small p-value means that it is unlikely that a pattern in the data at least as strong as the one seen would arise by chance, whereas a large p-value means that a pattern as strong as the one observed could easily arise by chance. It is still possible to occasionally get an apparently strong, but spurious relationship with a small p-value in the absence of a real effect, especially when conducting multiple tests. Conversely, especially when the data are noisy or confounding variables are not accounted for, it is possible for a real effect to be present despite the data having a pattern no more extreme than one that could be explained by chance alone (large p-value).

In Table 5.4.a through Table 5.4.g, the reported regression coefficients are based on z-score transformed abundances. This transformation scales each annual abundance estimates' deviation from the mean relative to typical variation around the mean abundance for the time-area combination under consideration. The regression coefficient gives the model predicted change in the demographic rate changes on the logit (fecundity, survival) or log (peanut-head) scale in response to an abundance change equal to one standard deviation in the annual abundance estimates for that time-area combination. This was intended to facilitate comparisons across regions that varied in both their mean abundance and typical degree of variability.

Table 5.4.a. Regression statistical summaries for the Council Coastwide EEZ aggregate area. The independent variable in each case is Chinook abundance after z-score transformation (*i.e.*, scaled so that 0 = mean and ±1 means 1 SD above or below the mean), and the "coefficient" gives the modeled effect on the log odds scale of a 1 SD change in abundance. Dependent variables are labeled under the column “Regression”. Lag effects described indicate the abundance variable used was either 1 (lag 1) or 2 (lag 2) years prior to the observed dependent variable.

Geographic Area	Timestep	Regression	Coefficient	Standard Error	p_Value
Coastwide (EEZ) aggregate	1 (October 1 – April 30)	Fecundity	0.1158	0.1208	0.3379
		Survival	0.0253	0.1198	0.8331
		Peanut	-0.2997	0.2810	0.2861
		Fecundity Lag 1	-0.0382	0.1244	0.7587
		Fecundity Lag 2	0.0377	0.1269	0.7663
		Survival Lag 1	0.2047	0.1265	0.1056
	2 (May 1 – June 30)	Fecundity	0.0762	0.1213	0.5299
		Survival	-0.0331	0.1180	0.7792
		Peanut	-0.1880	0.2687	0.4841
		Fecundity Lag 1	-0.0555	0.1247	0.6564
		Fecundity Lag 2	0.0299	0.1271	0.8143
		Survival Lag 1	0.1549	0.1244	0.2132
	3 (July 1 – September 30)	Fecundity	0.0913	0.1206	0.4492
		Survival	0.0114	0.1194	0.9239
		Peanut	-0.3185	0.2903	0.2725
		Fecundity Lag 1	-0.0447	0.1252	0.7211
		Fecundity Lag 2	0.0709	0.1256	0.5725
		Survival Lag 1	0.1828	0.1274	0.1513

Table 5.4.b. Regression statistical summaries for the North of Cape Falcon aggregate area.

Geographic Area	Timestep	Regression	Coefficient	Standard Error	p_Value
North of Falcon	1 (October 1 – April 30)	Fecundity	0.1272	0.1207	0.2918
		Survival	0.1433	0.1244	0.2490
		Peanut	-0.4807	0.3075	0.1180
		Fecundity Lag 1	0.0045	0.1238	0.9708
		Fecundity Lag 2	0.1112	0.1255	0.3760
		Survival Lag 1	0.2514	0.1295	0.0523
	2 (May 1 – June 30)	Fecundity	0.1051	0.1197	0.3802
		Survival	0.1053	0.1247	0.3985
		Peanut	-0.4419	0.3217	0.1696
		Fecundity Lag 1	-0.0103	0.1247	0.9343
		Fecundity Lag 2	0.1216	0.1228	0.3219
		Survival Lag 1	0.2028	0.1312	0.1223
	3 (July 1 – September 30)	Fecundity	0.0873	0.1199	0.4663
		Survival	0.1040	0.1255	0.4073
		Peanut	-0.4769	0.3372	0.1573
		Fecundity Lag 1	-0.0055	0.1248	0.9649
		Fecundity Lag 2	0.1398	0.1207	0.2469
		Survival Lag 1	0.1894	0.1321	0.1517

Table 5.4.c. Regression statistical summaries for the South of Cape Falcon aggregate area

Geographic Area	Timestep	Regression	Coefficient	Standard Error	p_Value
South of Falcon aggregate	1 (October 1 – April 30)	Fecundity	0.0966	0.1213	0.4257
		Survival	-0.0418	0.1181	0.7231
		Peanut	-0.1741	0.2657	0.5123
		Fecundity Lag 1	-0.0584	0.1244	0.6386
		Fecundity Lag 2	-0.0071	0.1279	0.9555
		Survival Lag 1	0.1543	0.1237	0.2122
	2 (May 1 – June 30)	Fecundity	0.0518	0.1217	0.6704
		Survival	-0.0846	0.1174	0.4710
		Peanut	-0.0786	0.2588	0.7615
		Fecundity Lag 1	-0.0669	0.1242	0.5903
		Fecundity Lag 2	-0.0130	0.1279	0.9191
		Survival Lag 1	0.1144	0.1225	0.3504
	3 (July 1 – September 30)	Fecundity	0.0753	0.1210	0.5338
		Survival	-0.0575	0.1176	0.6251
		Peanut	-0.1674	0.2673	0.5312
		Fecundity Lag 1	-0.0658	0.1249	0.5983
		Fecundity Lag 2	0.0049	0.1277	0.9691
		Survival Lag 1	0.1429	0.1245	0.2511

Table 5.4.d. Regression statistical summaries for the Salish Sea aggregate area.

Geographic Area	Timestep	Regression	Coefficient	Standard Error	p_Value
Salish Sea aggregate	1 (October 1 – April 30)	Fecundity	0.0320	0.1221	0.7931
		Survival	0.1343	0.1197	0.2620
		Peanut	-0.2926	0.2574	0.2557
		Fecundity Lag 1	-0.0504	0.1230	0.6821
		Fecundity Lag 2	0.1044	0.1280	0.4148
		Survival Lag 1	0.2216	0.1214	0.0680
	2 (May 1 – June 30)	Fecundity	0.0411	0.1223	0.7366
		Survival	0.1264	0.1204	0.2939
		Peanut	-0.3428	0.2635	0.1932
		Fecundity Lag 1	-0.0401	0.1234	0.7452
		Fecundity Lag 2	0.1020	0.1276	0.4240
		Survival Lag 1	0.2174	0.1227	0.0765
	3 (July 1 – September 30)	Fecundity	0.0335	0.1222	0.7843
		Survival	0.1217	0.1207	0.3135
		Peanut	-0.3043	0.2626	0.2466
		Fecundity Lag 1	-0.0430	0.1233	0.7271
		Fecundity Lag 2	0.0811	0.1275	0.5247
		Survival Lag 1	0.2109	0.1228	0.0858

Table 5.4.e. Regression statistical summaries for the South West / West Coast of Vancouver Island aggregate area.

Geographic Area	Timestep	Regression	Coefficient	Standard Error	p_Value
S.W. West Coast Vancouver Island coast	1 (October 1 – April 30)	Fecundity	0.0993	0.1203	0.4090
		Survival	0.1412	0.1260	0.2624
		Peanut	-0.4638	0.3169	0.1434
		Fecundity Lag 1	-0.0164	0.1248	0.8958
		Fecundity Lag 2	0.1318	0.1231	0.2842
		Survival Lag 1	0.2204	0.1313	0.0932
	2 (May 1 – June 30)	Fecundity	0.0961	0.1203	0.4242
		Survival	0.1331	0.1255	0.2891
		Peanut	-0.4592	0.3165	0.1468
		Fecundity Lag 1	-0.0204	0.1250	0.8703
		Fecundity Lag 2	0.1331	0.1229	0.2789
		Survival Lag 1	0.2079	0.1307	0.1116
	3 (July 1 – September 30)	Fecundity	0.0925	0.1208	0.4441
		Survival	0.2413	0.1290	0.0613
		Peanut	-0.4443	0.2982	0.1362
		Fecundity Lag 1	-0.0403	0.1252	0.7474
		Fecundity Lag 2	0.1513	0.1236	0.2207
		Survival Lag 1	0.2118	0.1288	0.1001

Table 5.4.f. Regression statistical summaries for the California Coast, south of Horse Mountain.

Geographic Area	Timestep	Regression	Coefficient	Standard Error	p_Value
California Coast	1 (October 1 – April 30)	Fecundity	0.0046	0.1221	0.9699
		Survival	-0.1016	0.1176	0.3876
		Peanut	-0.0055	0.2553	0.9828
		Fecundity Lag 1	-0.0642	0.1239	0.6045
		Fecundity Lag 2	-0.0218	0.1279	0.8649
		Survival Lag 1	0.0897	0.1218	0.4612
	2 (May 1 – June 30)	Fecundity	-0.0137	0.1222	0.9110
		Survival	-0.1181	0.1177	0.3153
		Peanut	0.0548	0.2537	0.8291
		Fecundity Lag 1	-0.0638	0.1238	0.6062
		Fecundity Lag 2	-0.0285	0.1279	0.8239
		Survival Lag 1	0.0634	0.1212	0.6007
	3 (July 1 – September 30)	Fecundity	-0.0370	0.1225	0.7627
		Survival	-0.0909	0.1178	0.4405
		Peanut	-0.0151	0.2559	0.9530
		Fecundity Lag 1	-0.0583	0.1243	0.6388
		Fecundity Lag 2	0.0148	0.1278	0.9076
		Survival Lag 1	0.0838	0.1226	0.4941

Table 5.4.g. Regression summaries for the Oregon coast south of Cape Falcon, including the California portion of the KMZ (CA-OR border through Horse Mountain).

Geographic Area	Timestep	Regression	Coefficient	Standard Error	p_Value
Oregon Coast	1 (October 1 – April 30)	Fecundity	0.1440	0.1208	0.2333
		Survival	-0.0021	0.1190	0.9861
		Peanut	-0.2734	0.2762	0.3222
		Fecundity Lag 1	-0.0508	0.1246	0.6834
		Fecundity Lag 2	0.0021	0.1277	0.9866
		Survival Lag 1	0.1826	0.1248	0.1433
	2 (May 1 – June 30)	Fecundity	0.1075	0.1211	0.3747
		Survival	-0.0457	0.1178	0.6979
		Peanut	-0.2060	0.2692	0.4442
		Fecundity Lag 1	-0.0638	0.1247	0.6088
		Fecundity Lag 2	0.0024	0.1277	0.9851
		Survival Lag 1	0.1526	0.1237	0.2176
	3 (July 1 – September 30)	Fecundity	0.1380	0.1206	0.2524
		Survival	-0.0302	0.1182	0.7986
		Peanut	-0.2551	0.2756	0.3545
		Fecundity Lag 1	-0.0634	0.1250	0.6121
		Fecundity Lag 2	-0.0018	0.1278	0.9888
		Survival Lag 1	0.1645	0.1246	0.1868

These results should be interpreted with caution. Nevertheless, in the majority of cases (70 percent; 88 of 126) the point estimates for the fitted relationships were of the expected sign (*i.e.* survival and fecundity increased with increasing Chinook salmon abundance while occurrence of peanut-head decreased with increasing Chinook salmon abundance). We use the term ‘expected’ here because our *a priori* expectation based on principles of ecology and physiology, and knowledge about the importance of Chinook salmon as prey, is for an increase in prey to have a neutral or positive effect on killer whale demography, and not a negative one. This was always the case for SRKW survival at a lag of one year, for SRKW survival based on current-year abundance estimates that excluded waters south of Cape Falcon, and for SRKW fecundity based on current-year abundances excluding waters south of Cape Falcon.

5.5 *Effects of Fisheries*

We estimated area-specific PFMC fishery removals in a two-step process. First, stock-specific removals were calculated across all fisheries for each modeled stock and each time step. This was to determine total stock abundance changes resulting from fishery removals. Then these removals were apportioned across space based on the assumed distribution of each stock (based on the spatial model and assumptions described in Section 5.1), rather than attempting to account for where fishery removals actually occurred and subsequent movement of fish within and across time steps.

The metric of abundance used to estimate area-specific PFMC fishery removals differs from the abundance metric used to quantify the relationship between Chinook salmon abundance and SRKW demographic metrics. The Workgroup agreed that the most appropriate abundance metric to use to examine relationships between Chinook salmon abundance and SRKW population metrics was starting abundance (pre-natural mortality, pre-fishery removals, pre-maturation in an individual time step). However, to estimate area-specific PFMC fishery removals, the Workgroup agreed that the most appropriate abundance metric to use was a starting abundance, with fishery mortalities from the season removed (hereafter referred to as a “post-fishery abundance”; this is distinct from abundance at the end of a time step, which would be further reduced by maturation and/or natural mortality). This decision was made because, while a starting cohort may be the best estimate of abundance at the start of a season, SRKW do not exclusively feed at the beginning of the season and feed throughout the time period examined. By examining a post-fishery abundance in the analysis, the group examined a maximum reduction estimate reflecting removals summed across the entire season. This likely overestimated the effect of within-season fishery removals in reducing prey available to SRKW. In part this is because removals are spread over the course of the season, so abundance will be higher earlier in the season before all of the removals have taken place. In addition, some unknown fraction of foregone removals early in the season would be expected to die of natural mortality (via causes other than SRKW predation) before the end of the season and thus not be available to SRKW. On the other hand, these calculations do not reflect cumulative effects of removals in earlier years.

For FRAM stocks (excluding SRFC), we estimated PFMC area removals of each stock by comparing the post-fishery abundances calculated for each timestep by the FRAM "Validation" run (with fisheries as they occurred) with the post-fishery abundances calculated for the start of each timestep in a "Zero PFMC" FRAM run where all PFMC area ocean salmon fisheries were zeroed out, but other salmon fisheries outside PFMC jurisdiction were modeled as before. PFMC removals were calculated based on the difference between the corresponding "Zero PFMC" and "Validation" abundances.

For Upriver Columbia Springs, we assumed no removals in PFMC area ocean salmon fisheries, based on the extremely low rates of recovery of CWTs for this stock.

For RRFC, KRFC, and SRFC we calculated September 1 abundances each year as described in Section 5.1, and then calculated "Zero PFMC" October 1 abundances for each year by adding back September 1 ocean harvest (assumed to all occur within PFMC areas, which is likely appropriate because these stocks are very rarely encountered further north) and discounting for one month of assumed natural mortality. "Zero PFMC" abundances for later timesteps were calculated by discounting the "Zero PFMC" October 1 abundances by assumed natural mortality over the corresponding months elapsed. PFMC removals were calculated based on the differences between corresponding starting abundances calculated in Section 5.4 and the "Zero PFMC" abundances described here.

Estimated Changes in Predicted Vital Rates Due to Effects of Removals on Chinook Abundance

Due to the different abundance metrics used in this analysis, the starting abundance regressions could not be directly applied to post-fishery abundance data calculated to assess the effects on prey availability from the fisheries. In order to estimate mean changes in the predicted demographic metrics due to the removal of Chinook salmon from the PFMC fisheries, all regressions performed in regression analysis were rerun, using the same methodology as described in Section 5.4, but using post-fishery abundance data (post-season runs) rather than starting abundance data.

Post-season and "zero PFMC" abundances for each year and spatio-temporal box were used in conjunction with regressions rerun using post-fishery abundance data to create point estimates of SRKW survival (lag 0 years, 1 year), fecundity (lag 0 years, 1 year, 2 years), and the occurrence of peanut head. Differences in SRKW population metrics derived from point estimates in the post-season and "zero PFMC" runs were used to assess yearly changes in SRKW population metrics that would be predicted to have occurred due to fishing mortality. For lag effect estimates, SRKW population parameter changes were not available for 1992 (lag 1 and lag 2 effects) or 1993 (lag 2 effects) because Chinook post-fishing abundance estimates were not available prior to 1992.

Table 5.5.a. Mean estimates of change in survival (lag 0, lag 1), fecundity (lag 0, lag 1, and lag 2), and occurrence of peanut head across the series of years available in the analysis. Time steps 1, 2, and 3 represent "October through April", "May through June", and "July through September", respectively. Annual changes used in the mean represent predicted SRKW metrics from the post-season runs subtracted from the "zero PFMC" run. Survival is expressed as an annual change in survival rate (positive values indicate increase in the absence of fishing) for young females. Fecundity is expressed as an annual change fecundity rate (positive values indicate increase in the absence of fishing) for age 20 females. Occurrence of peanut head represents the change in the annual number of predicted peanut heads (negative values indicate decrease in the absence of fishing).

Area	TimeStep	Survival	Fecundity	Peanut Head	Fecundity L 1	Survival L 1	Fecundity L 2
CALI	1	0.0%	0.0%	0.00	-0.1%	0.0%	0.0%
CALI	2	-0.1%	-0.1%	-0.01	-0.7%	0.1%	0.2%
CALI	3	-0.5%	-0.5%	-0.07	-1.8%	0.4%	2.2%
COASTWIDE	1	0.0%	0.2%	-0.01	-0.1%	0.0%	0.1%
COASTWIDE	2	0.0%	0.5%	-0.03	-0.4%	0.2%	0.4%
COASTWIDE	3	0.0%	1.2%	-0.06	-0.8%	0.3%	1.2%
NOF	1	0.0%	0.1%	0.00	0.0%	0.0%	0.1%
NOF	2	0.0%	0.2%	-0.01	0.0%	0.1%	0.3%
NOF	3	0.0%	0.2%	-0.02	-0.1%	0.1%	0.4%
OR	1	0.0%	0.2%	-0.01	-0.1%	0.0%	0.0%
OR	2	0.0%	0.6%	-0.02	-0.3%	0.1%	0.1%
OR	3	-0.1%	1.6%	-0.04	-0.8%	0.2%	0.1%
SALISH	1	0.0%	0.0%	0.00	0.0%	0.0%	0.0%
SALISH	2	0.0%	0.0%	-0.01	-0.1%	0.0%	0.1%
SALISH	3	0.0%	0.1%	-0.01	-0.2%	0.1%	0.1%
SOF	1	0.0%	0.2%	0.00	-0.1%	0.0%	0.0%
SOF	2	-0.1%	0.5%	-0.02	-0.5%	0.2%	0.2%
SOF	3	-0.2%	1.7%	-0.06	-1.4%	0.3%	0.6%
SWWCVI	1	0.0%	0.0%	0.00	0.0%	0.0%	0.0%
SWWCVI	2	0.0%	0.1%	-0.01	0.0%	0.0%	0.2%
SWWCVI	3	0.1%	0.2%	-0.01	-0.2%	0.1%	0.4%

The code used for the analysis is publicly available here:
https://github.com/dappdrd/PFMC_SRKW/blob/master/Harvest_analysis.R

5.6 Key Uncertainties

The analyses undertaken for evaluating effects of PFMC fisheries on the prey base of killer whales are largely similar to those reviewed by Hilborn *et al.* (2012), with additional details in Ward *et al.* (2013). Thus, most of the same caveats and uncertainties about these models that have been described in Hilborn *et al.* (2012) also apply here. Among the summaries by Hilborn *et al.* (2012) were that "considerable caution is warranted in interpreting the correlative results as confirming a linear causal relationship between Chinook salmon abundance and SRKW vital rates". These relationships are likely non-linear, the relationships may be influenced by small sample sizes of killer whale births and deaths, and the relationships may arise from uncertainties in the indices of Chinook abundance used for fisheries management. Additionally, the Hilborn *et al.* (2012) panel cautioned that there are "many potential reasons why all foregone Chinook salmon catch would not be available to SRKW". Thus, even if all ocean fisheries were closed, only a fraction of those removals would be made available to killer whales. These assumptions and limitations identified by Hilborn *et al.* (2013), as well as additional limitations and uncertainties, are addressed in more detail below.

Statistical model assumptions

The models assume that the effect of Chinook salmon abundance in a particular season and area is the same every year (*i.e.* assume stationarity), and the same for all pods, regardless of where SRKW actually spent the most time that year, and do not account for any variation at finer spatial or temporal scales than those defined by the model. The logistic regressions used for survival and fecundity assume that all whales of the same age (fecundity) or sex/stage (survival) have identical probabilities of giving birth or dying in a given year, ignoring individual variability (aside from excluding whales who gave birth the prior year from the fecundity analysis). The logistic regression model assumes that survival or fecundity on the logit scale is a linear function of Chinook abundance. The Poisson GLM used for incidence of peanut-head assumes that all whales have the same probability of displaying peanut-head in a given year, and that this rate, on the logarithmic scale, is a linear function of Chinook salmon abundance. The Poisson model further assumes that the variance in the number of SRKW displaying peanut head in a given year is equal to the expectation for the number of SRKW displaying peanut-head that year. Both the logistic regressions and the Poisson GLM assume that time/area-specific Chinook abundance is measured without error. Unaccounted-for measurement error with constant variance in the independent (putative "driver") variable in a simple linear regression is known to bias estimated coefficients toward zero ("attenuation bias" or "regression dilution"), but the effects are harder to characterize for more complicated models (Chesher 1991).

Uncertainty in Chinook salmon stock abundances

The uncertainty associated with Chinook salmon abundance forecasts in general is relatively well appreciated, but there is also substantial uncertainty in retrospective abundance estimates. Harvest and escapement estimates are themselves uncertain, but ocean abundance estimates depend further on unverified assumptions about natural mortality, constant adult natural mortality rates across years, mortality associated with fish caught but released, drop-off mortality, and bycatch mortality in other fisheries that are not accounted for in the management models.

Additionally the FRAM uses a "base period" to estimate fishing mortalities by stock, age, fishery, and time step. The current Chinook FRAM base period is represented by coded wire tag recoveries from fishing years 2007–2013. If stock distributions differ considerably from the 2007–2013 base period, stock-specific or if tagged fish are not representative of untagged fish (*e.g.,* hatchery versus wild differences), fishery mortality estimates from the model reflect reality less well.

The effects of fishery removals on the availability of Chinook as potential SRKW prey depends on patterns in natural mortality, and how many fish from potentially foregone harvest would die from natural causes (other than SRKW predation) rather than remain available as prey. As Hilborn *et al.* (2012) note, natural mortality likely varies across years, due in part to the relative abundance of Chinook salmon and their multiple predators. However, nearly all models used in Chinook management, including the ones used here, assume constant adult natural mortality (but see Allen *et al.* 2017). Assumptions about natural mortality and when it is applied will change estimates of "foregone removals" that are actually available to SRKW as food. If natural mortality is higher than assumed, the models will overestimate the ability of foregone harvest to increase Chinook abundance. On the other hand, the models used did not consider the effects of fishery impacts on age-2 fish, nor did it consider multi-year effects (i.e. fishery removals in prior years can reduce the abundance of older fish in the current year, and fishery removals in the current year can reduce the abundance of older fish in future years).

Uncertainty in Chinook stock distributions

The Shelton *et al.* (2019) distribution model is subject to uncertainty due to sampling error in harvest data, assumptions about natural mortality, assumptions about how catch per unit effort scales with local abundance (and the consistency of metrics of fishing effort across time and space), the assumption that stocks have the same spatial distribution every year, and the assumption that a subset of marked hatchery releases are representative of all releases from the corresponding stock and also representative of the natural-origin component of those stocks (an assumption made in FRAM as well). The model published by Shelton *et al.* (2019) does not include data through 2016 as we used here, however, estimated distribution from the period used by Shelton *et al.* (brood years 1977 – -1990) may be more precise because of higher sampling rates. Work is in progress to account for inter-annual variability in the Shelton *et al.* model, and to incorporate GSI information from both hatchery- and natural-origin fish, but no results were available in time to inform this analysis.

Additionally, a temporal mismatch exists between the Shelton *et al.*, 2019 model and FRAM. FRAM abundances are based on three different time steps, corresponding to Winter (October through April), Early Summer (May through June), and Late Summer (July through September). However, time steps in Shelton *et al.*, 2019 are offset by a month relative to the FRAM model, with Winter designated as November–May, Early Summer designated as June–July, and Late Summer designated as August–October. Although this mismatch causes a disconnect between the two models, the Shelton *et al.*, 2019 model is believed by the workgroup to be the better model to characterize Chinook distribution, and future work will be explored to produce results from the Shelton model that are compatible with FRAM time steps.

Finally, the spatial model ignores changes in Chinook salmon spatial distribution within each timestep, and assumes that the effects on Chinook salmon abundance from fishery removals are distributed across space in proportion to Chinook salmon abundance, rather than based on where fishery removals actually occur and how quickly fish redistribute themselves across space.

Lack of information on Chinook distributions during winter

The model used to apportion Chinook abundance through space (Shelton *et al.* 2019) depends on coded-wire tag recoveries from ocean fisheries directly targeting Chinook salmon. Effort in these fisheries has been very limited or nonexistent in winter and early spring for most years because fisheries do not currently occur at these times (with several exceptions, including the 4B treaty troll fishery in Washington State near Neah Bay). Efforts are underway to include additional data sources (*e.g.*, from salmon bycatch in trawl fisheries) to learn more about Chinook spatial distributions in the winter and early spring, but no results were available in time to inform this analysis.

Limited information on distribution for most spring-run Chinook stocks

Quantitative distribution estimates from Shelton *et al.* (2019) were only available for fall-run stocks. Efforts are underway to extend this model to spring-run stocks, but the generally lower catch rates and resultant smaller sample sizes for these stocks pose a challenge. Ongoing efforts to share information across coded-wire tag, genetic stock identification, and trawl bycatch datasets should increase the statistical power and provide better insights about spring run distributions, although the seemingly more offshore distribution of some spring run stocks will pose an ongoing challenge to models based on fishery-dependent data. These results will have to be modeled at a coarser spatial resolution for instance, compared to fall stocks, because of significantly smaller sample sizes.

Effects of changes in Chinook salmon size and age structure

The utility of Chinook salmon as prey depends on more than their abundance alone. Older Chinook salmon are larger and thus provide more nutrition per fish than younger fish. In addition, Chinook salmon that mature at younger ages spend less time in the ocean and thus spend less time potentially available as prey, possibly meaning less food for SRKW per smolt entering the ocean. At the same time, returning spawners per smolt may be higher for younger fish that experience less cumulative mortality risk, potentially increasing the availability of Chinook salmon prey per smolt for SRKW specifically targeting aggregations of returning spawners near river mouths. It appears that both hatchery- and natural-origin Chinook salmon are becoming smaller and maturing more rapidly throughout most of the Pacific coast (Ohlberger *et al.* 2018). The workgroup did not analyze any impacts of these changes in size at age during modeling exercises. These changes introduce additional uncertainties in quantifying the biomass of prey available to the whales, as well as the relative selectivity across stocks that differ in their energy content (O’Neill *et al.* 2014).

Uncertainty in the distribution of SRKW

Much of the knowledge of SRKW distribution is based on sightings reported in the inland waters of the Salish Sea, especially in summer months (Olson *et al.* 2018; Hauser *et al.* 2006). The distribution of SRKW year to year can be characterized as variable, and possibly subject to short term trends. Over the last several years, for example, many social groups of the SRKW population have not spent much time in inland waters during the summer relative to their historical occurrence (Olson *et al.* 2018). For non-summer months, sighting data is generally limited. Several satellite tags have been deployed on SRKWs in winter months to characterize the winter distribution (Jan - Apr). Data from these deployments suggests that J pod has a distribution in the Salish Sea, concentrated in the northern Strait of Georgia and western entrance to the Strait of Juan de Fuca (Hanson *et al.* 2018). However, J pod tag data is limited to an extremely small sample size (one tag deployed in February 2012 for three days; one tag deployed in December 2013 for 31 days; one tag deployed December 2014 for 49 days; Hanson *et al.* 2018) and additional data on the distribution of J pod during the winter would be beneficial. K and L pods are estimated to have a more frequent coastal distribution, with a winter/spring concentration off the Columbia River, and Washington coast (Hanson *et al.* 2018). Distribution in spring and fall months has been characterized from acoustic recorders (Hanson *et al.* 2013) and additional analyses are being conducted to update these estimates.

Differential responses to changes in Chinook abundance among pods

In the winter, J pod appears to remain much more within the Salish Sea relative to K and L pods that spend more time in coastal waters, thus it is likely that they would have differential responses to changes in the abundance of particular Chinook stocks compared to K and L. However, considerable statistical power is lost when analyzing one pod at a time due to lower sample sizes. As a result the workgroup has opted to continue to examine all three pods together.

Uncertainty in the factors driving changes in the distribution of SRKW

Other than factors related to prey abundance, or phenology, it is unclear what factors may influence SRKW distribution. Some have speculated that changes in the age structure of SRKW (particularly the loss of older animals) may alter future distributions, if historical knowledge is lost. It is unclear to what degree SRKWs or other killer whales actively avoid vessels, or other populations of killer whales, however both of these may also influence distribution.

Uncertainty in the ability of SRKW to switch to alternative prey sources

The degree to which killer whales are able to or willing to switch to non-preferred prey sources (*i.e.*, prey other than Chinook salmon) is also largely unknown, and likely variable depending on the time and location. We do not account for varying abundance and availability of alternative prey sources in these analyses. Previous genetics work has suggested that SRKWs switch from Chinook to other salmon in fall months (particularly coho and chum salmon, Ford *et al.* 2016). Though a small number of samples have been collected, fecal samples collected in winter suggest a diet that is still more than 50% Chinook, but also includes contributions from groundfish (halibut, lingcod) and steelhead (Hanson *et al.* 2018). In addition to small sample sizes, the spatial location of these recent samples is confounded with season (*e.g.* few summer diet samples have been collected outside of the Salish Sea, and few winter diet samples have been collected in the Salish Sea). Diet data reflecting longer integration windows (bulk stable isotopes) have been analyzed recently, and suggest that year to year variability may affect diet variation (*e.g.* Chinook salmon consumption may be higher when they are more abundant, and lower in years when coastwide abundance is low; Warlick *et al.* in review).

Patterns of temporal variation in competing threats

A number of threats unrelated to Chinook abundance have been identified as potential threats to SRKW. These include, but are not limited to: additional anthropogenic threats (contaminants in the food web, increased noise levels around vessels, risks of ship strikes, potential effects of oil spills,), disease, ecosystem effects (competition from other populations of fish-eating killer whales, and other marine mammals including seals and sea lions), inherent risks associated with small populations (inbreeding depression, demographic stochasticity, skewed sex ratios at birth with unknown causes), and behavioral risks (infanticide, Allee effects). To the extent that these factors vary across years, they will confound the effects of changes in Chinook salmon abundance, but they can only be included as model covariates if annual measurements are available, which by and large they are not.

Chinook salmon stocks whose abundances are not included in the modeling

North of Cape Falcon, non-modeled stocks include those north of Vancouver Island, Hupp Springs, Washington Coastal Springs, and Tsoo-Yess Falls. Many of these stocks are relatively small in magnitude (*e.g.,* Hupp Springs, Washington Coastal Springs, Tsoo-Yess) or are present primarily outside of the core SRKW assessment area (*e.g.,* stocks north of Vancouver Island).

South of Cape Falcon, it is likely that the two most important non-modeled stocks are Klamath-Trinity spring run (for which 1992-2016 adult river run sizes were on median 21 percent as large as the river run size of Klamath River Fall Chinook salmon) and California Coastal Chinook salmon (for which 0.23 genetically-identified fish were found for every 1 genetically-identified Klamath River Chinook during sampling of California recreational fisheries in 1998-2002 [Satterthwaite *et al.* 2015]). Rogue River Spring and Central Valley Spring Chinook might also be of particular value to SRKW due to their river return timing coincident with presence of SRKW in southern waters, but their run sizes are relatively small, with typical river run sizes less than 10 percent of the typical river run sizes of Klamath River Fall Chinook and Sacramento River Fall Chinook salmon, respectively. See Appendix A for further details on non-modeled stocks.

6 INTEGRATION AND SYNTHESIS

[Pending]

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APPENDIX A

Chinook salmon stocks excluded from the Assessment

The following describes stocks which are known to occur in the EEZ, but for which the Council either does not currently utilize models to account for these stocks, or in the specific case of Sacramento Winter Chinook, a model is available but the stock's contribution to potential SRKW prey base was considered insubstantial. Although their abundance and distribution may affect SRKWs demographics, the Workgroup here provides the rationale for exclusion of these stocks:

- Sacramento Winter Chinook – Sacramento Winter Chinook escapement as a percentage of SRFC escapement had a median value of 1.3 percent for 1992-2016 (for this and the other Central Valley stock comparisons, 1992-2000 escapements were obtained from the CHINOOKPROD data set, obtained from the US Fish and Wildlife Service's Anadromous Fish Restoration Program [http://www.fws.gov/stockton/afrp, downloaded March 2011] and 2001-2016 escapements were obtained from PFMC 2019a). Sacramento Winter Chinook also have small body sizes, a primarily age-3 maturation rate, and have ocean distributions heavily concentrated south of Point Arena, CA (O'Farrell *et al.* 2012), all of which suggests they are unlikely to make substantial contributions as SRKW prey.

- Central Valley Spring Chinook – Central Valley Spring Chinook escapement as a percentage of SRFC escapement had a median value of 4.6 percent for 1992-2016. Note that the estimated Central Valley Spring Chinook escapement does not include spring run fish spawning in natural areas on the Feather River, which are included in the fall run escapement estimate and thus contribute to the SI modeled in Council fisheries.

- Other components of the Central Valley Fall Chinook Stock Complex (San Joaquin Fall and Sacramento Late-Fall Chinook) – Together escapement of these two as percentage of SRFC escapement had a median value of 6.4 percent for 1992-2016.

- Klamath River Spring Chinook – Adult river run size for Klamath River Spring Chinook as a percentage of adult river run size for KRFC had a median value of 21 percent for 1992-2016 (Klamath River Spring Chinook data from CDFW's "Current – 2017 Spring Chinook Megatable 1-Mar-2019.xlsx", KRFC data from PMFC 2019).

- California Coastal Chinook – Abundance of this stock is not well characterized. Genetic stock identification (GSI) sampling of California recreational ocean fisheries from 1998-2002 (Satterthwaite *et al.* 2015) suggested that 0.23 California Coastal Chinook were caught for each Klamath River Chinook (fall or spring run).

- Smith River Chinook – Abundance of this stock is not well characterized, but a few unpublished estimates suggest annual escapements on the order of 16,000 fish (Shelton *et al.* 2019), less than 20% of the median KRFC adult river run size for 1992-2016.

- Rogue River Spring Chinook – Terminal river returns are under 10,000 fish in most years (C. Kern ODFW pers. comm.), so mostly under 10 percent of the median KRFC adult river run size for 1992-2016.

- Other Southern Oregon Chinook stocks outside the Rogue River – Myers et al (1998) states that Rogue River fish are numerically dominant among these stocks.

Overall, we deemed it unlikely that excluding these less abundant stocks (all of which, with the exception of Sacramento Winter Chinook, lack vetted models for generating ocean abundance estimates, even retrospectively) would substantially affect the conclusions of later analyses relating SRKW performance to aggregate Chinook abundance. Further, again with the exception of Sacramento Winter Chinook, we do not have vetted abundance forecasts available for the excluded southern stocks, so we would have no way of evaluating their expected contribution to the SRKW prey base during pre-season planning. Relative catch rates from genetic stock identification studies might be informative on relative ocean abundance for similarly distributed stocks, but sample sizes and spatio-temporal coverage are currently limited. Relative escapements or river run sizes might provide some indication of relative ocean abundances, but are confounded by differences in age structure, maturation schedules, natural mortality, and ocean fishing mortality.

Appendix A References

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APPENDIX B

Abundance models for southern stocks

For SRFC, we used a modification of the Sacramento Index (SI, O'Farrell *et al.* 2013) to characterize adult (ages 3 and older combined) ocean abundances through time. The SI is the sum of adult river run size and ocean harvest of SRFC south of Cape Falcon and serves to index abundance on September 1 of each management year (management years south of Cape Falcon run from September 1 to August 31). Note that the SI does not account for natural mortality, nor does it account for unharvested immature fish remaining in the ocean for another year, so it likely underestimates pre-season ocean abundance. While we were not able to account for immature fish remaining in the ocean, we made new calculations that incorporate natural mortality. We assumed monthly adult natural mortality of m=0.0184, equivalent to 20 percent annual mortality. We then calculated August 1 ocean abundance N_8 as $N_8=R/(1-m)+H_8$ where R represents adult river run size and H_8 is adult ocean harvest of SRFC in August. For earlier months, $N_t=N_{t+1}/(1-m)+H_t$ (and for management years, month 12 precedes month 1). Our October 1 abundances do not match the SI values reported in PFMC 2019 Table II-1 both because our calculation reflects removals during September and because we adjust numbers upward throughout the year to account for natural mortality.

For KRFC, we used monthly age-specific (ages 3 and older) ocean abundance estimates produced by cohort reconstructions informing the Klamath Ocean Harvest Model (KOHM, Mohr 2006; September 1 values for ages 3 and 4 are available in PFMC 2019 Table II-3). Ratios between monthly age-specific abundance estimates in the KRFC cohort reconstruction reflect the combined effects of fisheries removals and assumed values of natural mortality.

For RRFC, we characterized age-specific September 1 ocean abundances using the ROPI (ROPI, PFMC 2019 Table II-7). The ROPI is calculated based on age-specific RRFC river run size, scaled up by age-specific ocean harvest rates estimated for KRFC and assumed natural mortality. Therefore, we assumed that age-specific values of RRFC abundance for later months would have the same ratio to the ROPI that monthly age-specific abundances for KRFC have to their corresponding September 1 estimates.

SRKW appear most likely to be present in waters south of Cape Falcon during the winter and early spring (Hanson *et al.* 2018). Thus, fishery removals of Chinook salmon during October could affect prey availability when SRKW are most likely to be present (ocean fisheries are closed during the winter). For SRFC, a maximum of three percent of the SI was harvested during October during the years 1992-2016, with annual median and mean of 0.9 percent and one percent, respectively. For KRFC, total reduction in adult abundance between October 1 and November 1 (reflecting both fisheries and assumed natural mortality) ranged from four to five percent with median five percent. Thus, it appears unlikely that accounting for October fishery removals would substantially change the results of later analyses.

References for Appendix B

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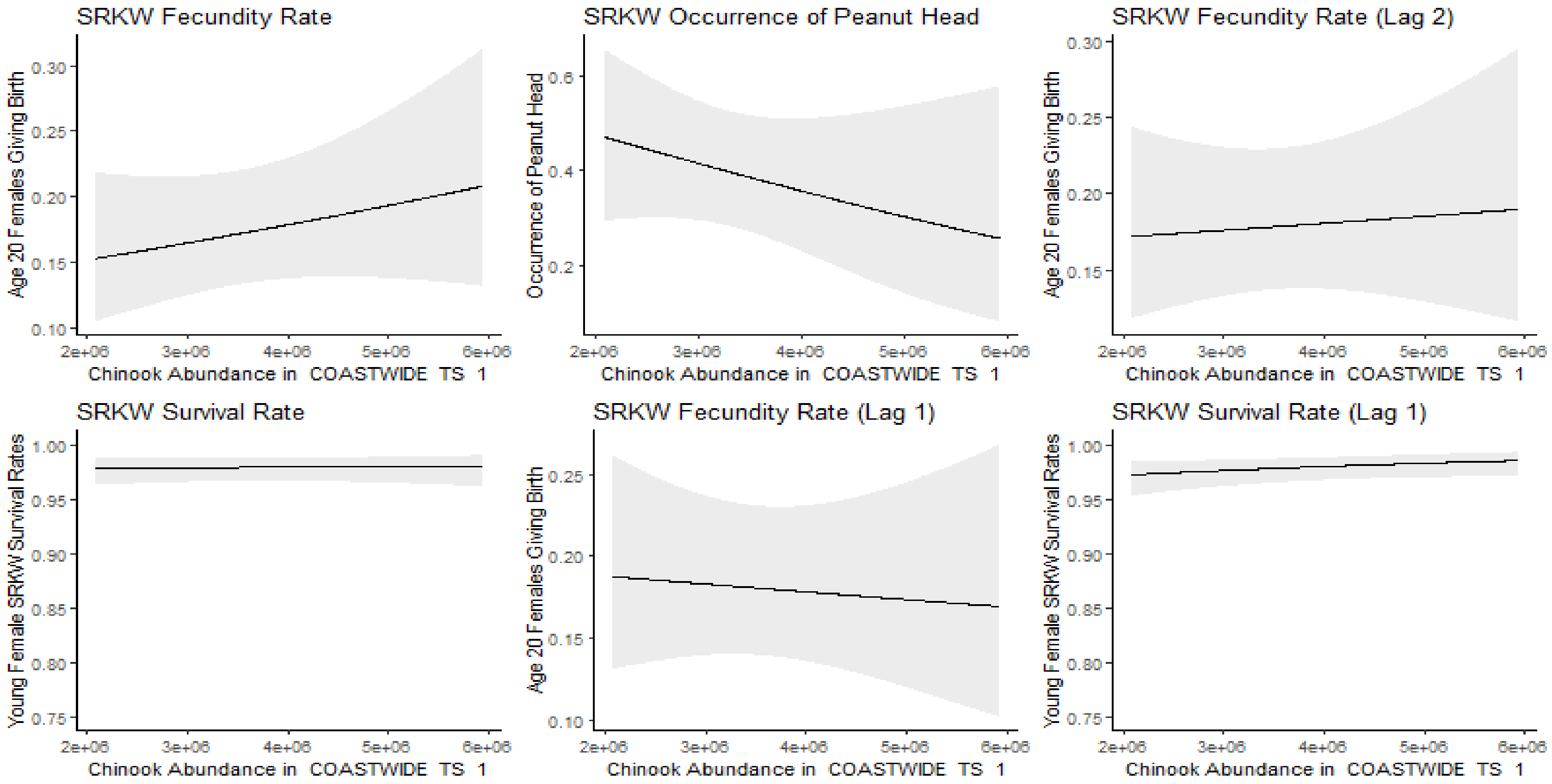
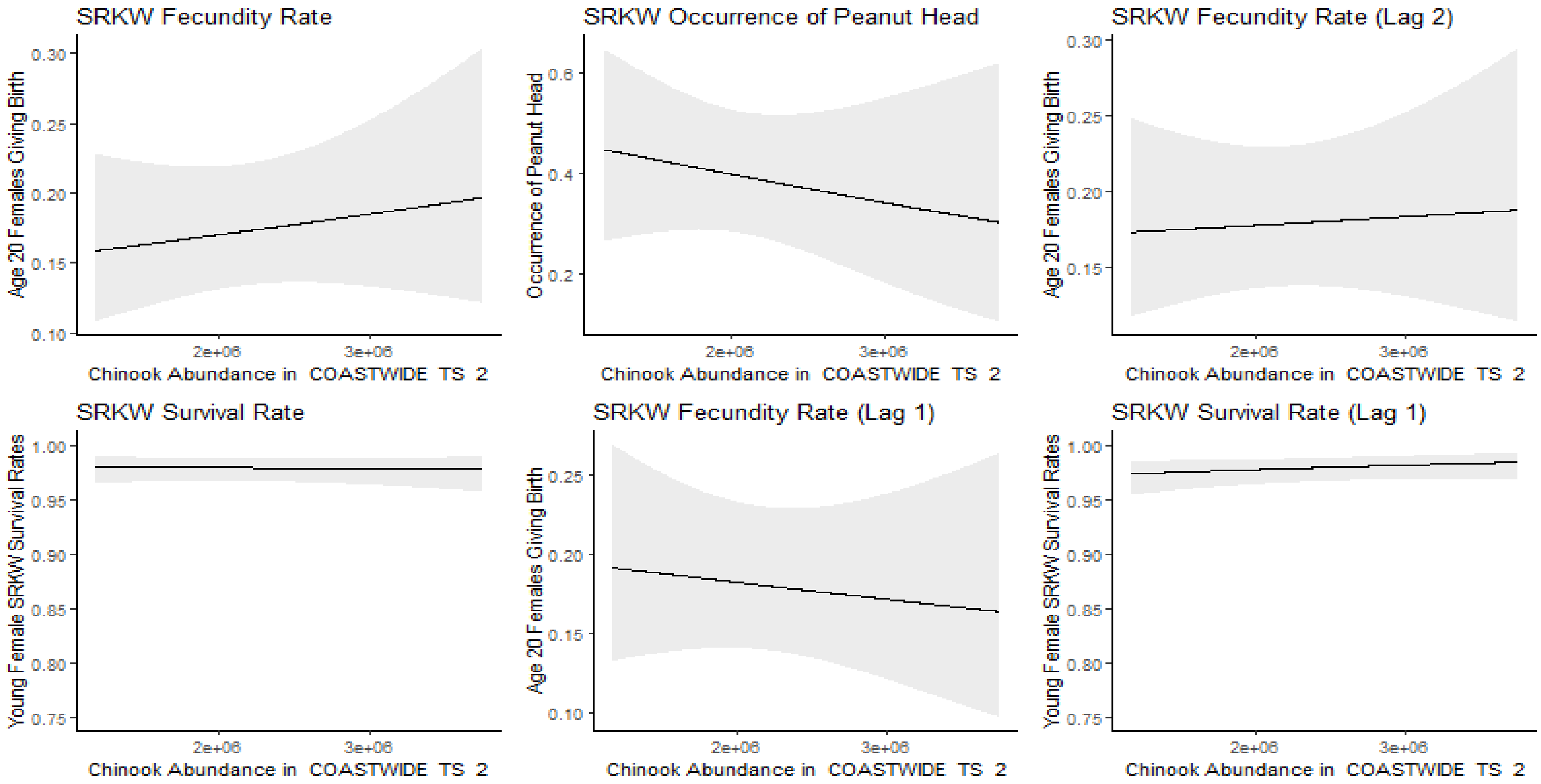
APPENDIX C

Regression model outputs

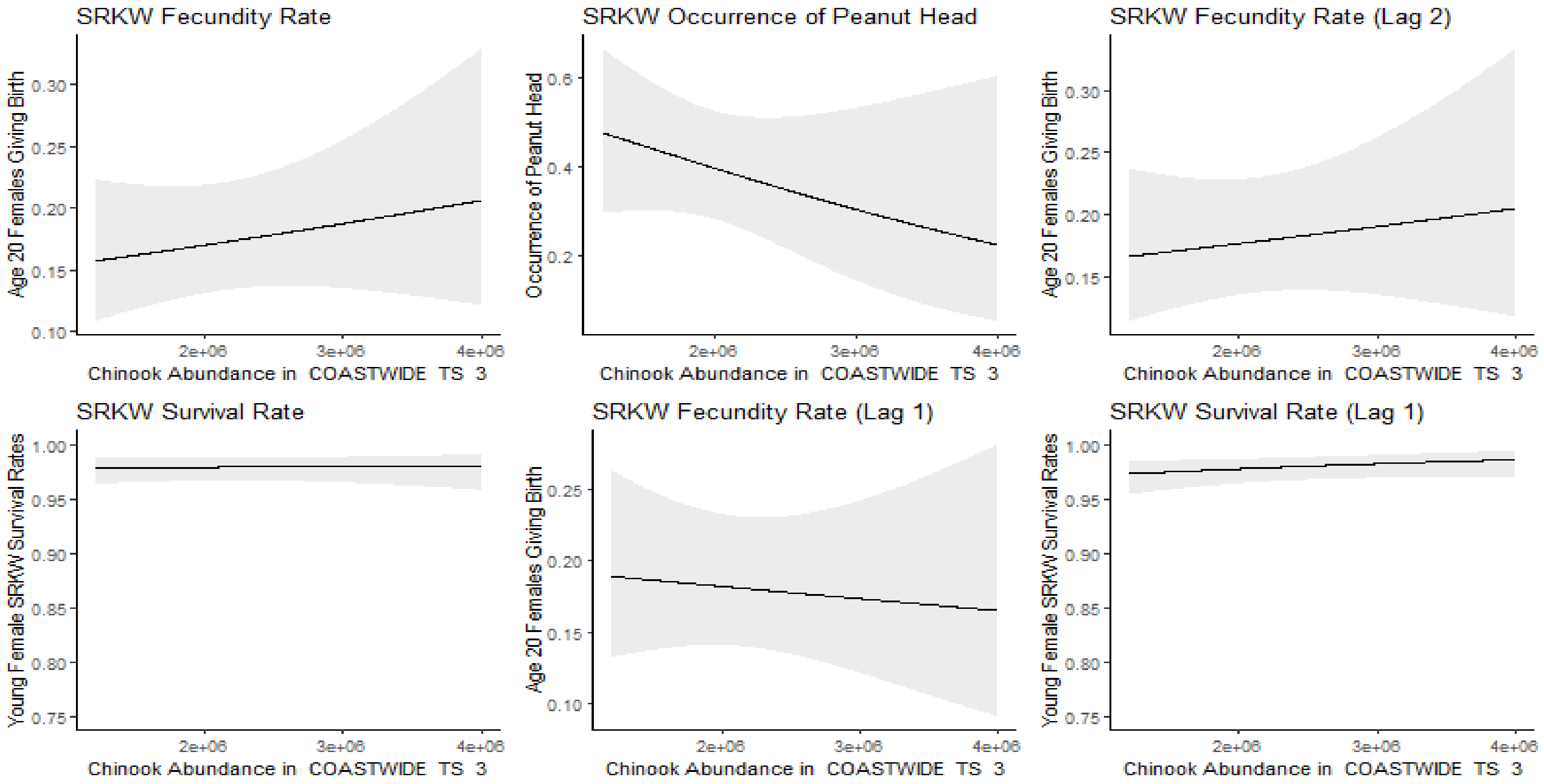
Appendix C Figure 1. Demographic rates modeled as functions of Coastwide (EEZ) aggregate abundance in Timestep 1. Figures illustrate fecundity with no lag (top left), survival with no lag (bottom left), peanut head with no lag (top middle), fecundity with 1 year lag (bottom middle), fecundity with 2 year lag (top right), and survival with 1 year lag (bottom right).



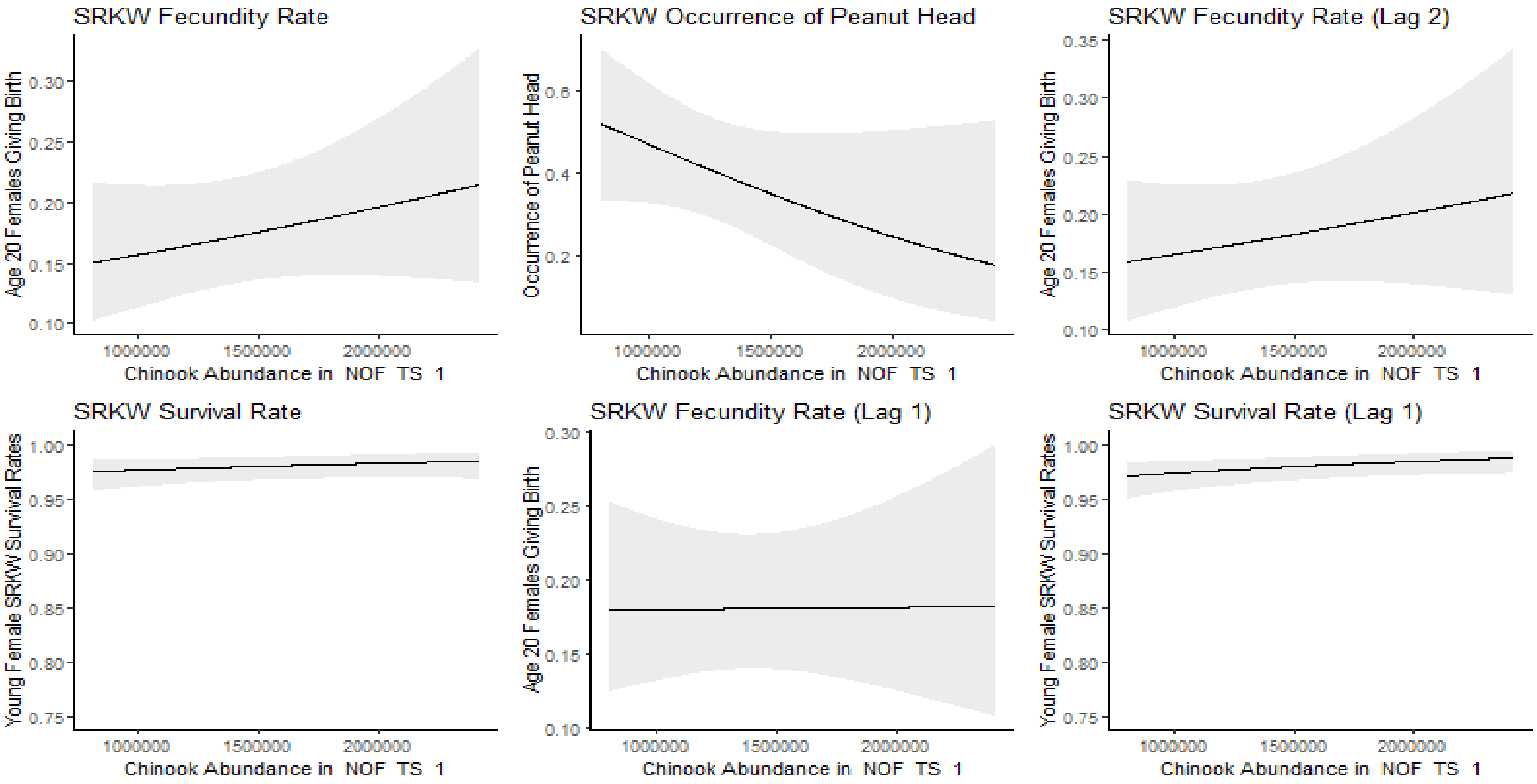
Appendix C Figure 2. Coastwide (EEZ) aggregate Timestep 2



Appendix C Figure 3. Coastwide (EEZ) aggregate Timestep 3



Appendix C Figure 4. North of Falcon Timestep 1



SRKW Fecundity Rate

SRKW Occurrence of Peanut Head

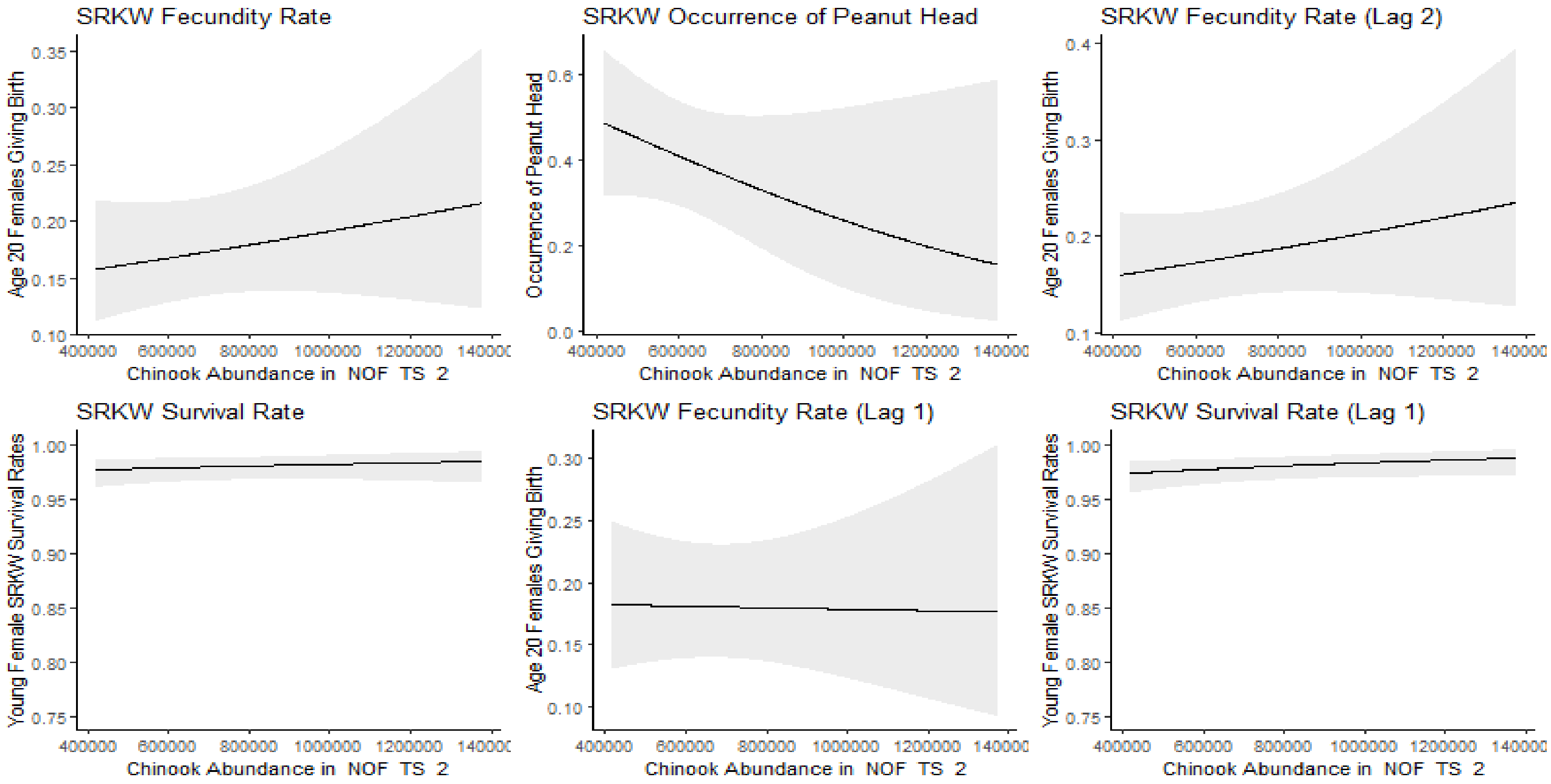
SRKW Fecundity Rate (Lag 2)

SRKW Survival Rate

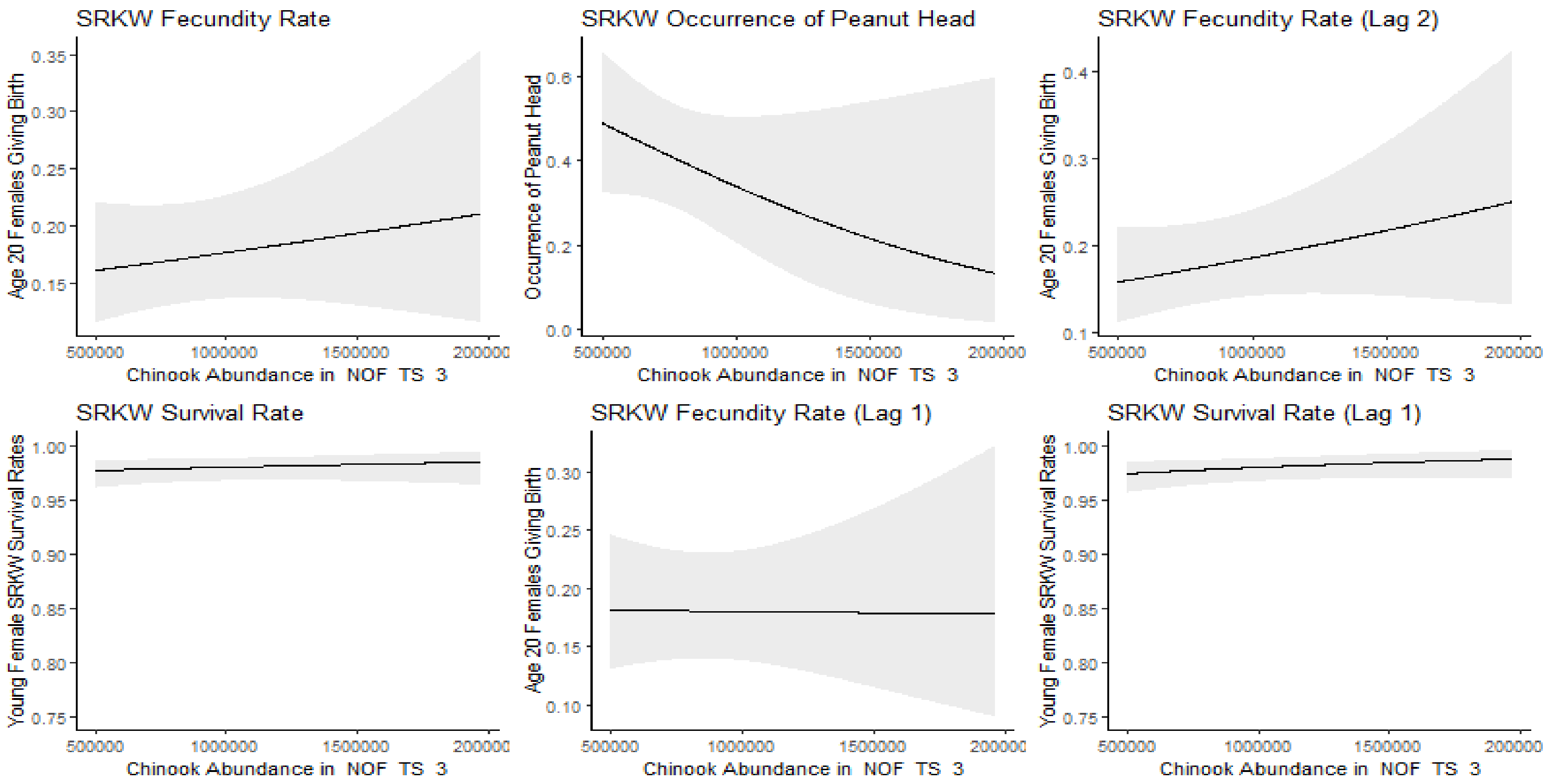
SRKW Fecundity Rate (Lag 1)

SRKW Survival Rate (Lag 1)

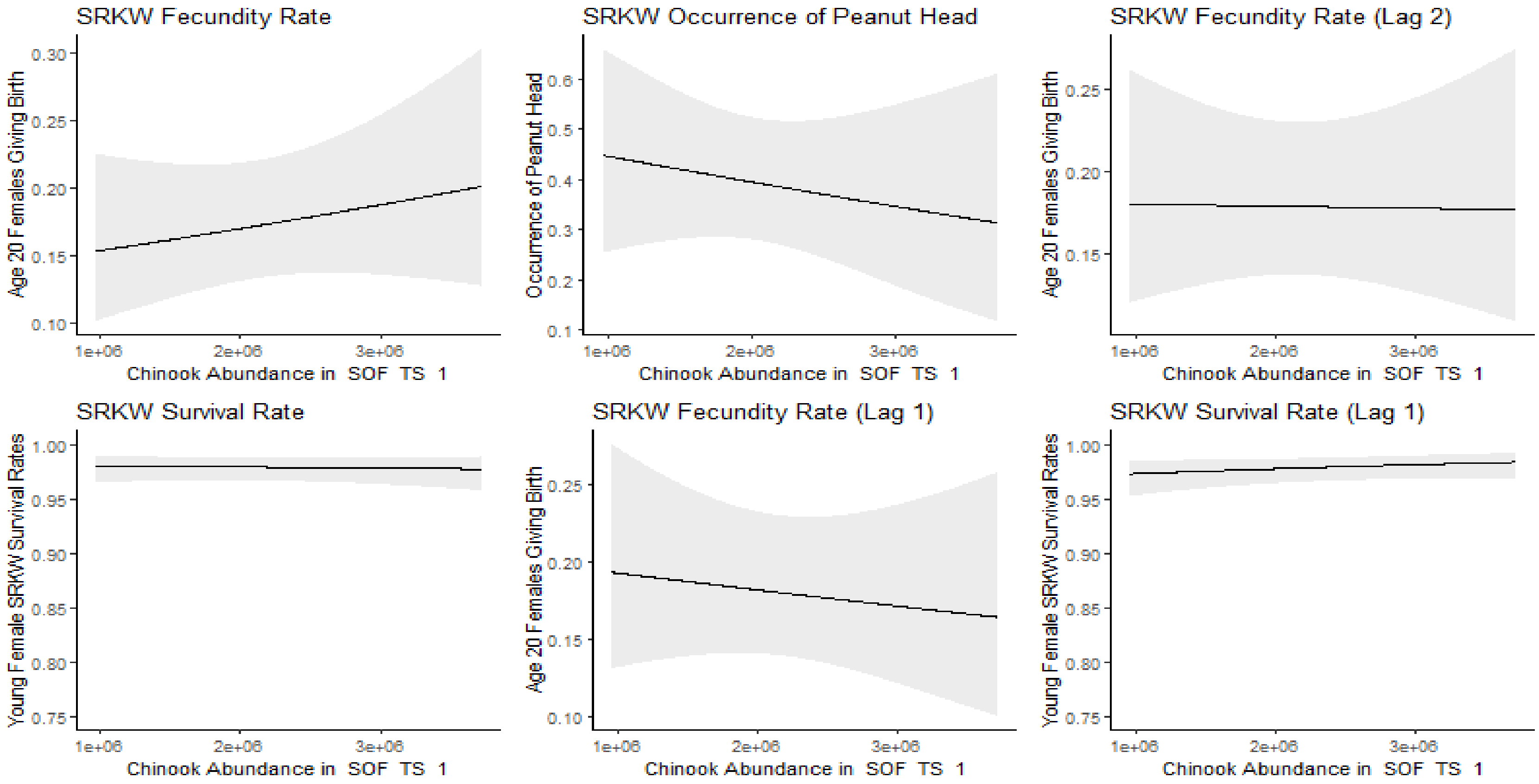
Appendix C Figure 5. North of Falcon Timestep 2



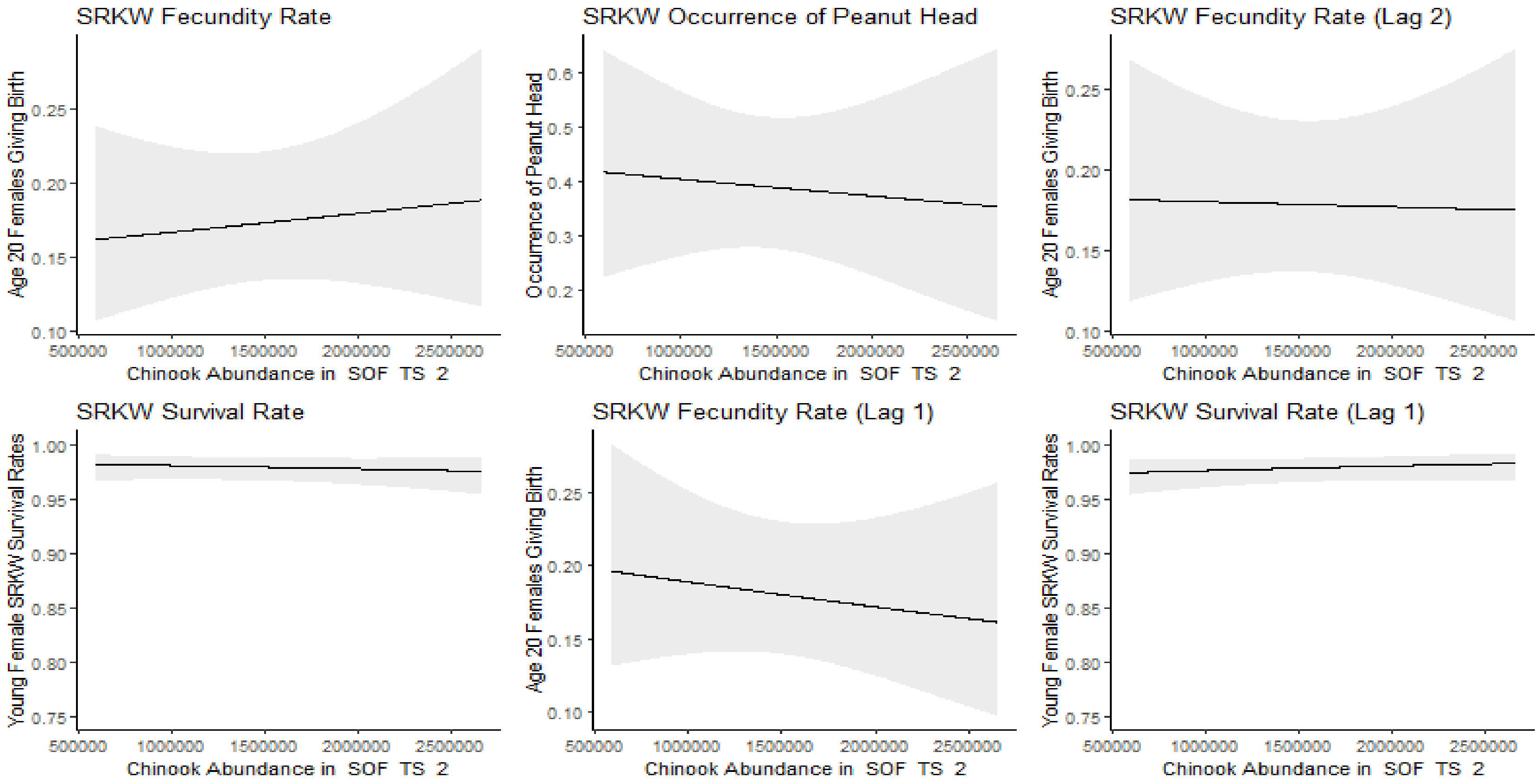
Appendix C Figure 6. North of Falcon Timestep 3



Appendix C Figure 7. South of Falcon Timestep 1



Appendix C Figure 8. South of Falcon Timestep 2


SRKW Fecundity Rate

SRKW Occurrence of Peanut Head

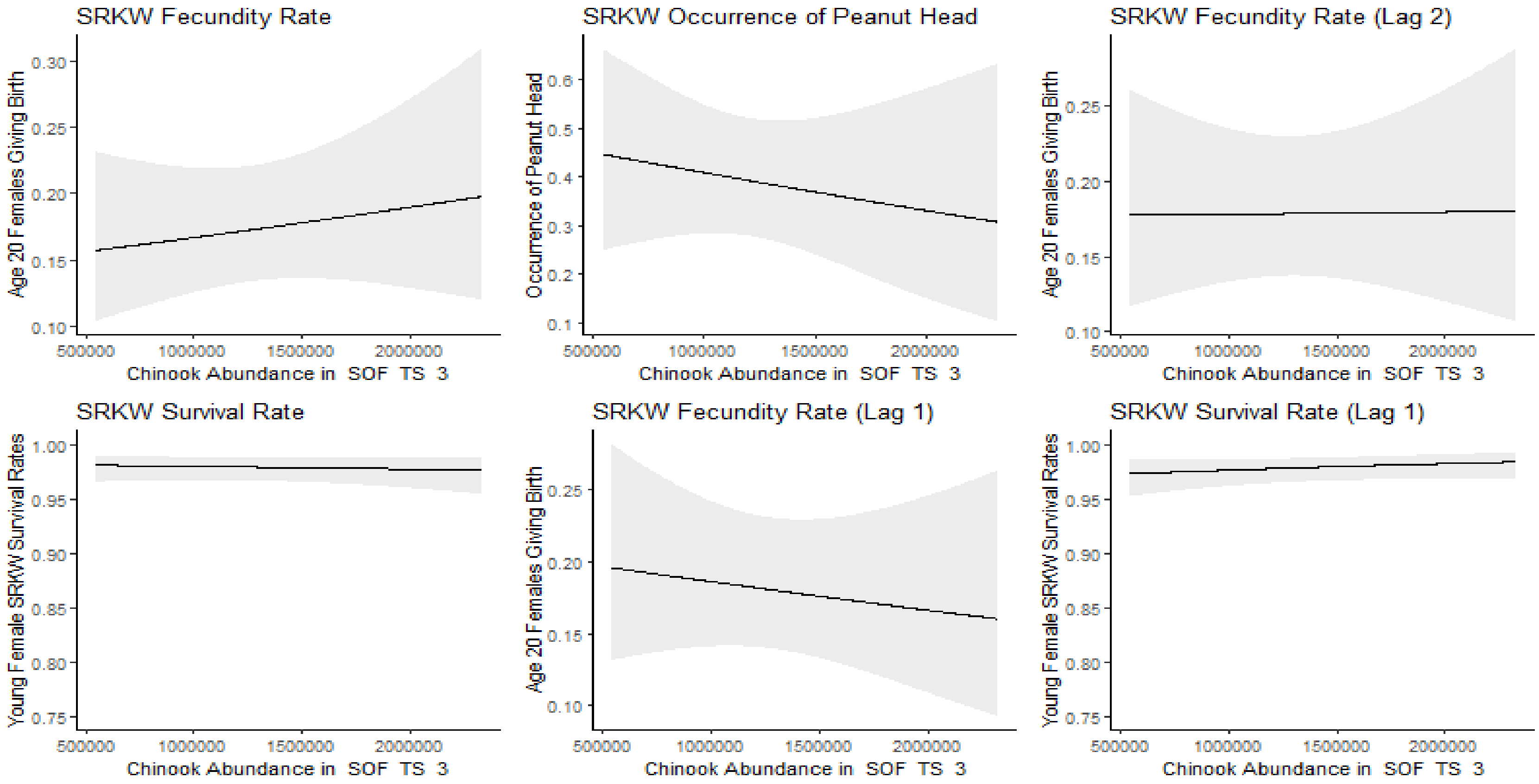
SRKW Fecundity Rate (Lag 2)

SRKW Survival Rate

SRKW Fecundity Rate (Lag 1)

SRKW Survival Rate (Lag 1)

Appendix C Figure 9. South of Falcon Timestep 3


SRKW Fecundity Rate

SRKW Occurrence of Peanut Head

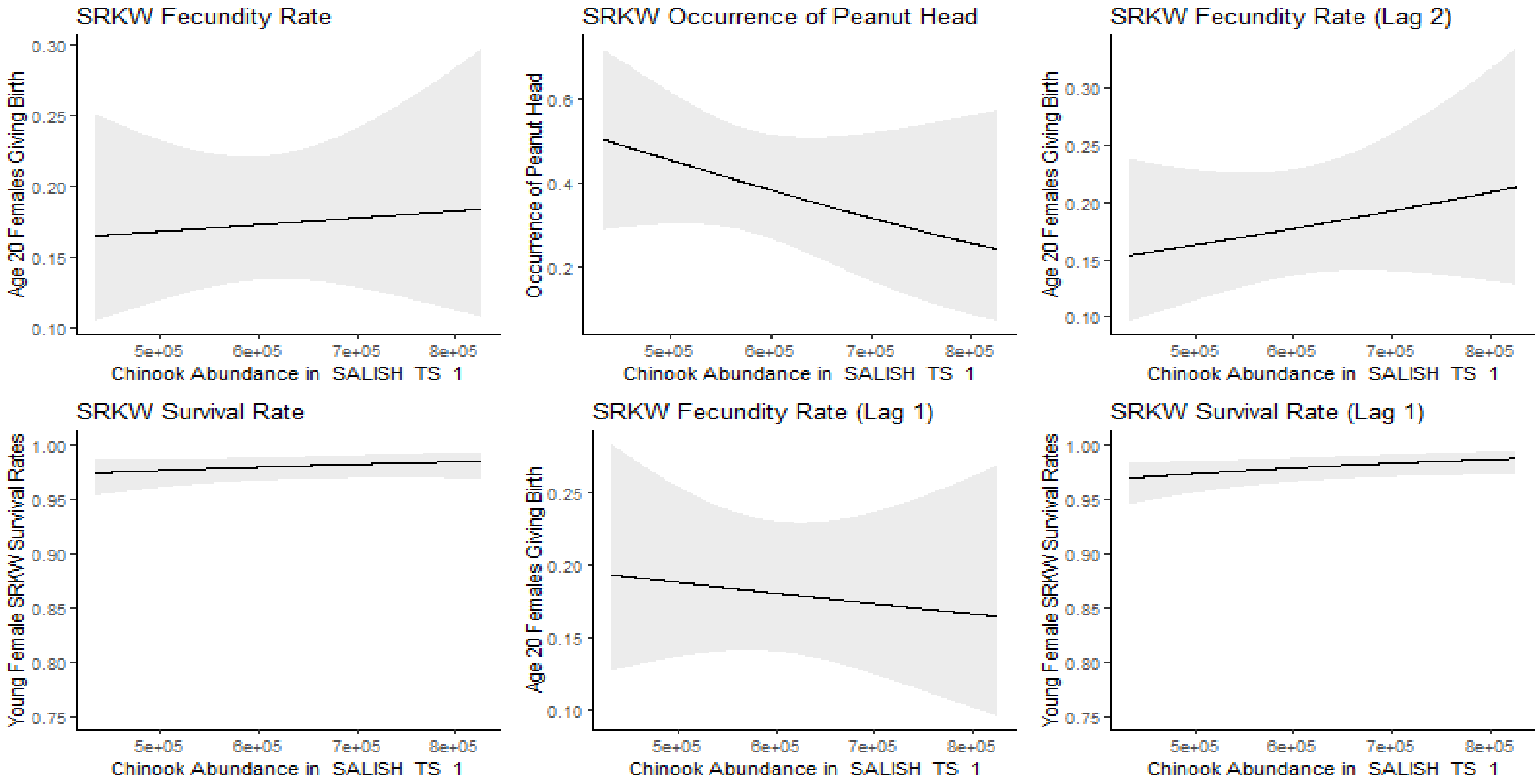
SRKW Fecundity Rate (Lag 2)

SRKW Survival Rate

SRKW Fecundity Rate (Lag 1)

SRKW Survival Rate (Lag 1)

Appendix C Figure 10. Salish Sea aggregate Timestep 1


SRKW Fecundity Rate

SRKW Occurrence of Peanut Head

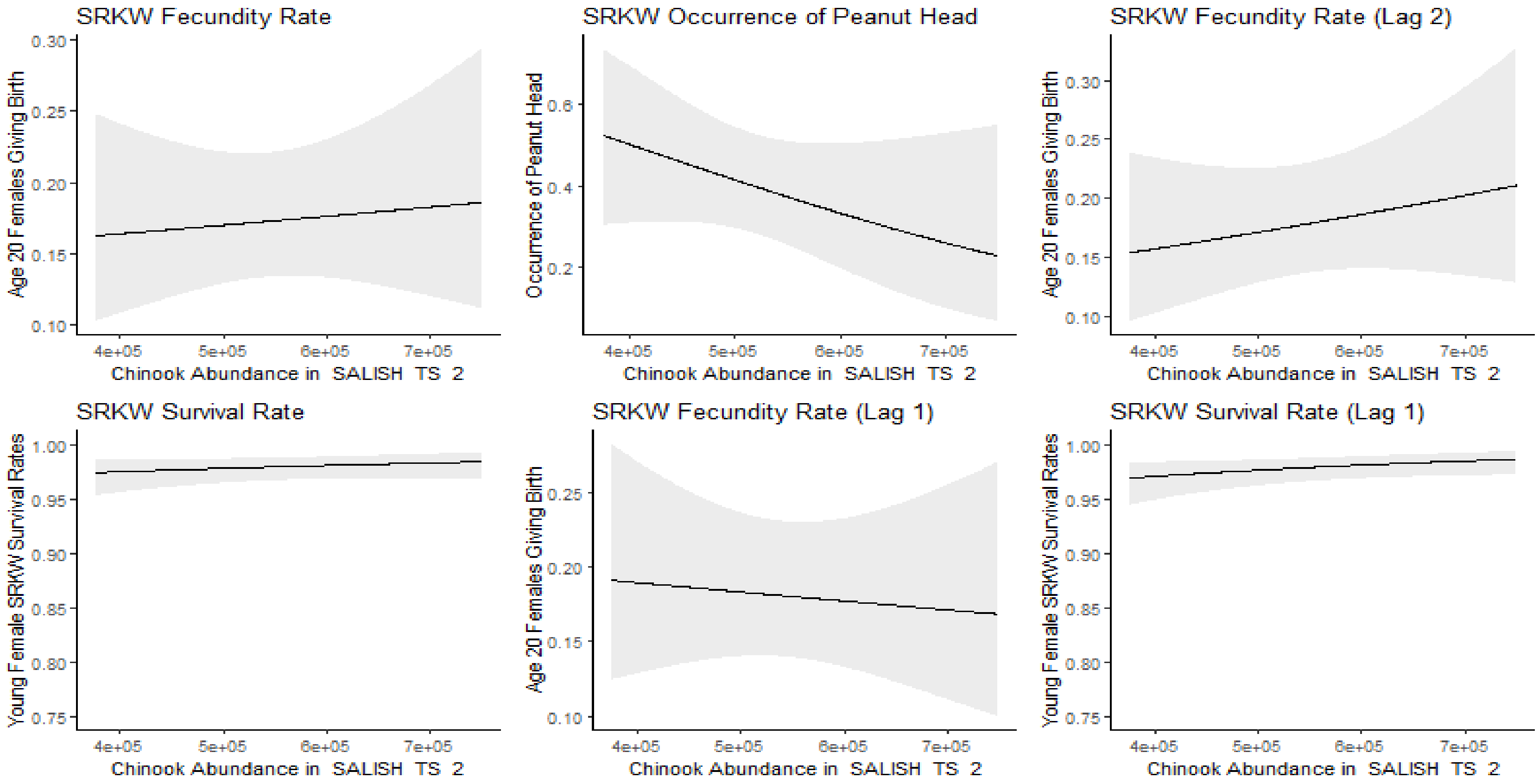
SRKW Fecundity Rate (Lag 2)

SRKW Survival Rate

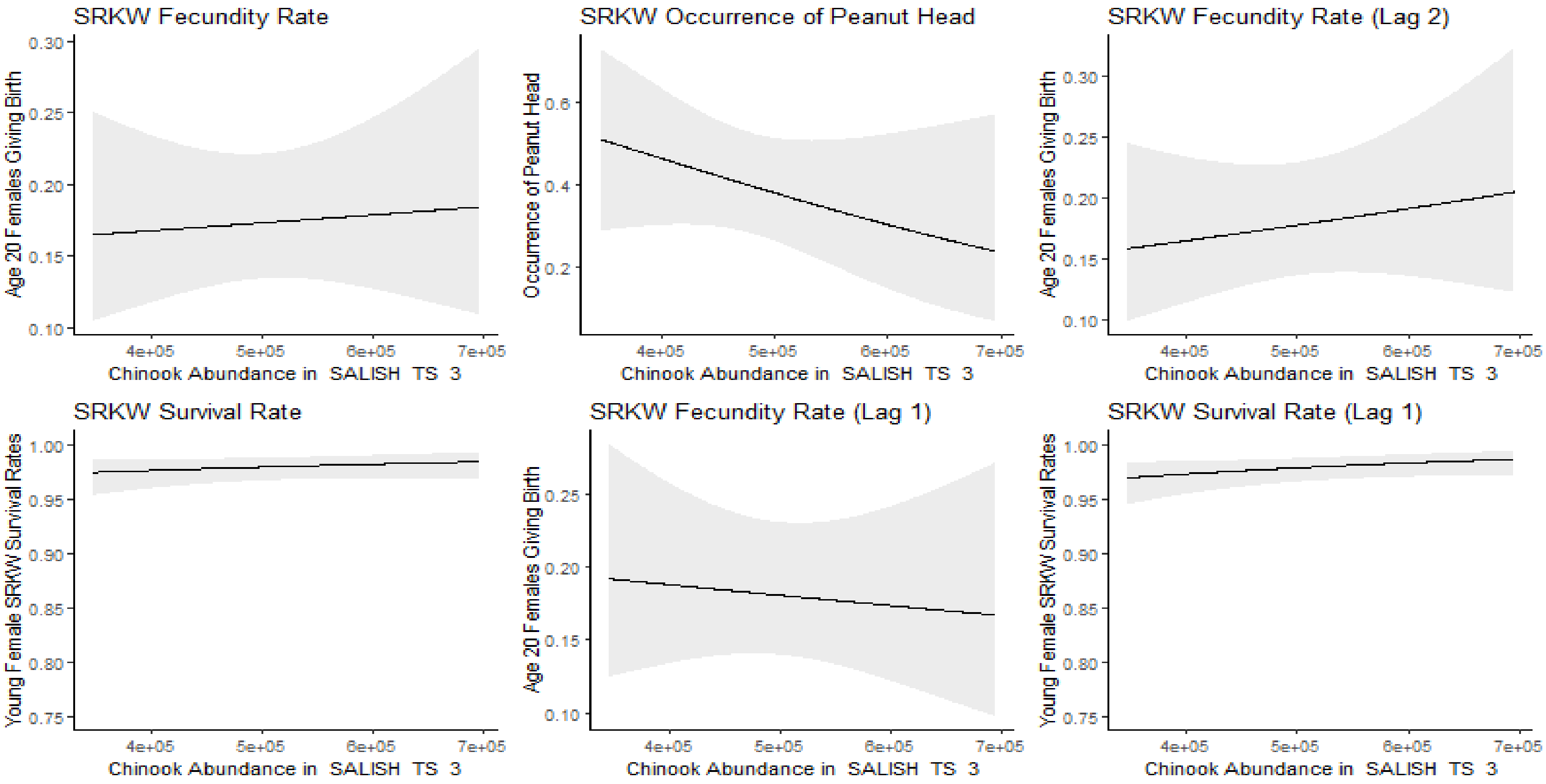
SRKW Fecundity Rate (Lag 1)

SRKW Survival Rate (Lag 1)

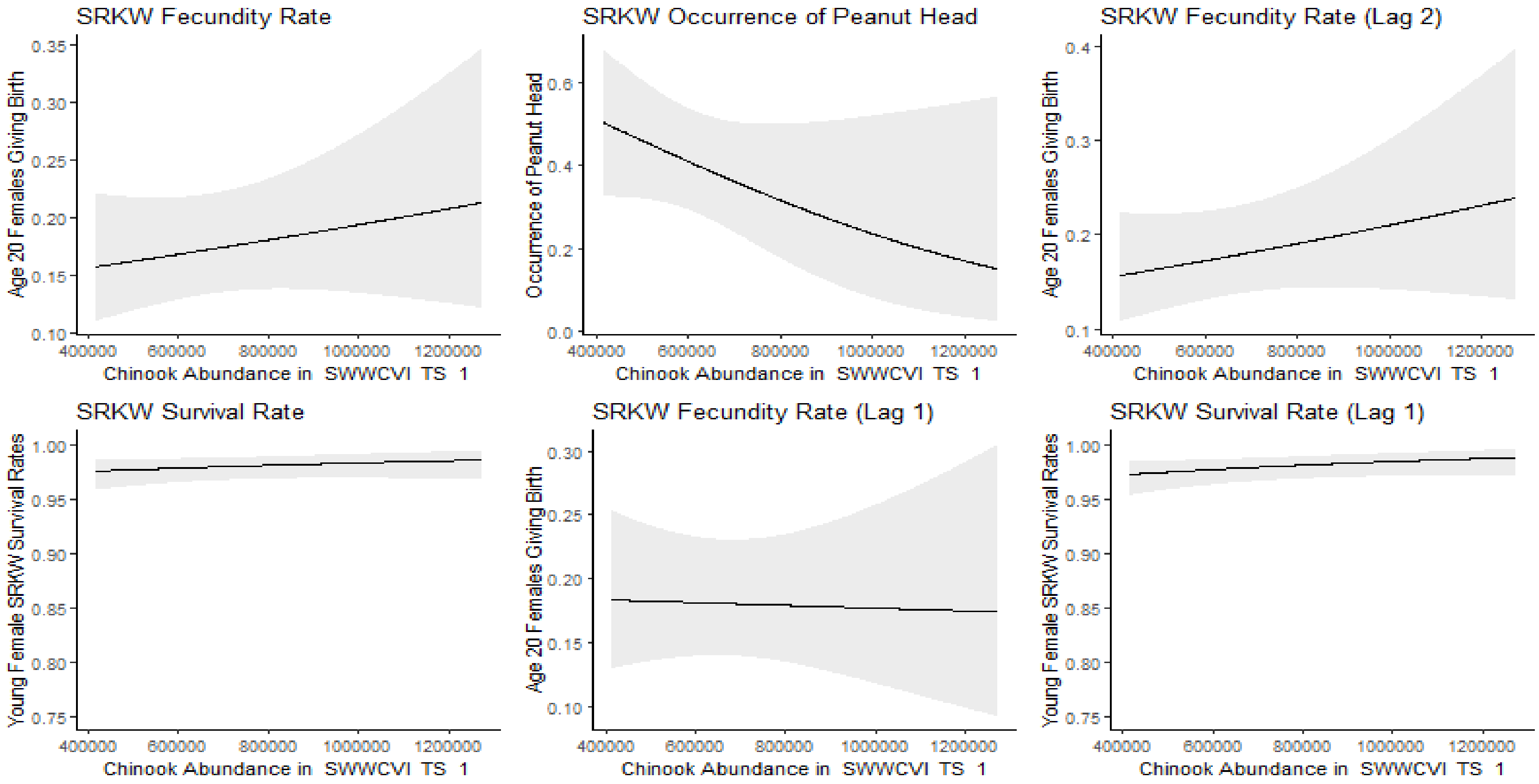
Appendix C Figure 11. Salish Sea aggregate Timestep 2



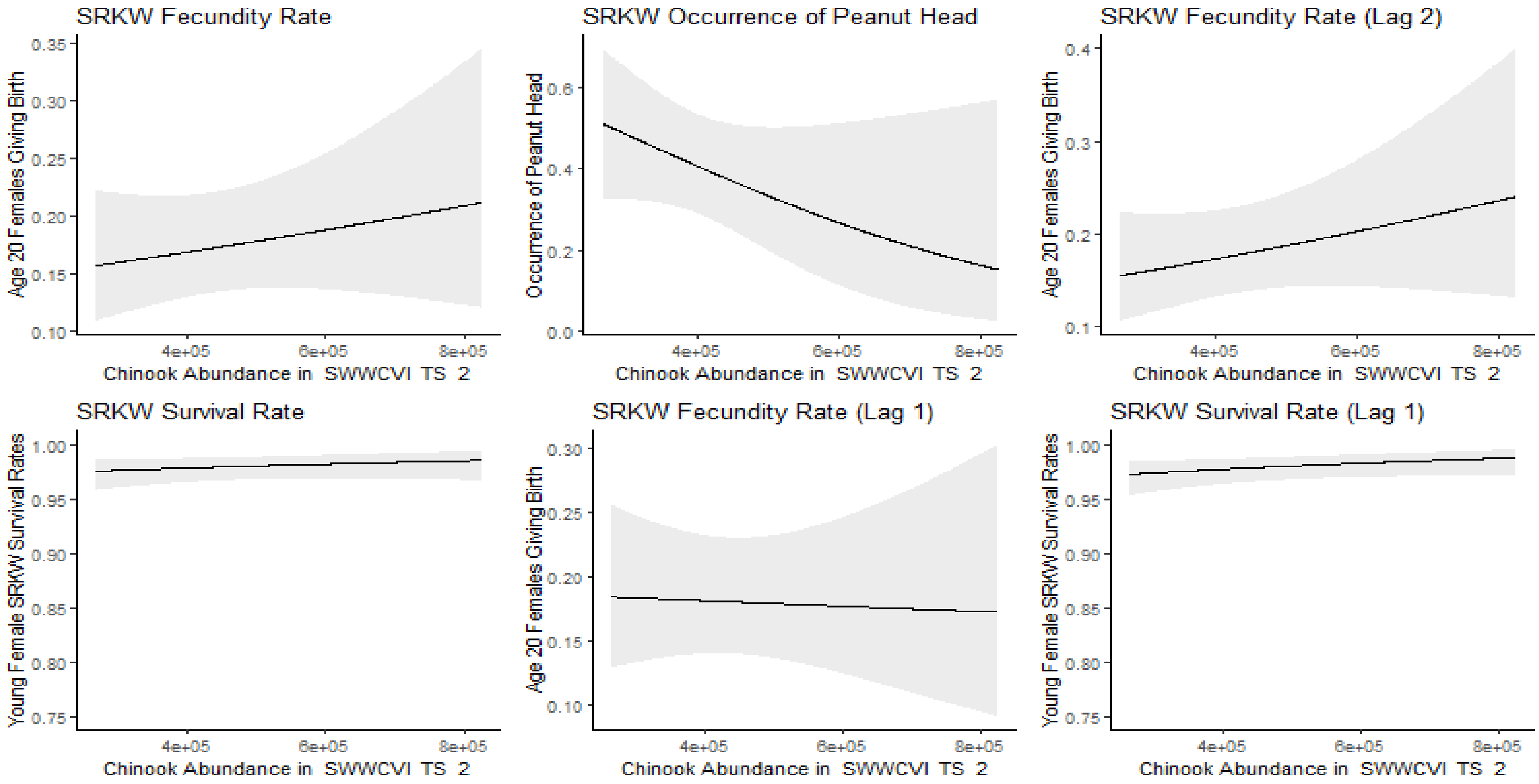
Appendix C Figure 12. Salish Sea aggregate Timestep 3



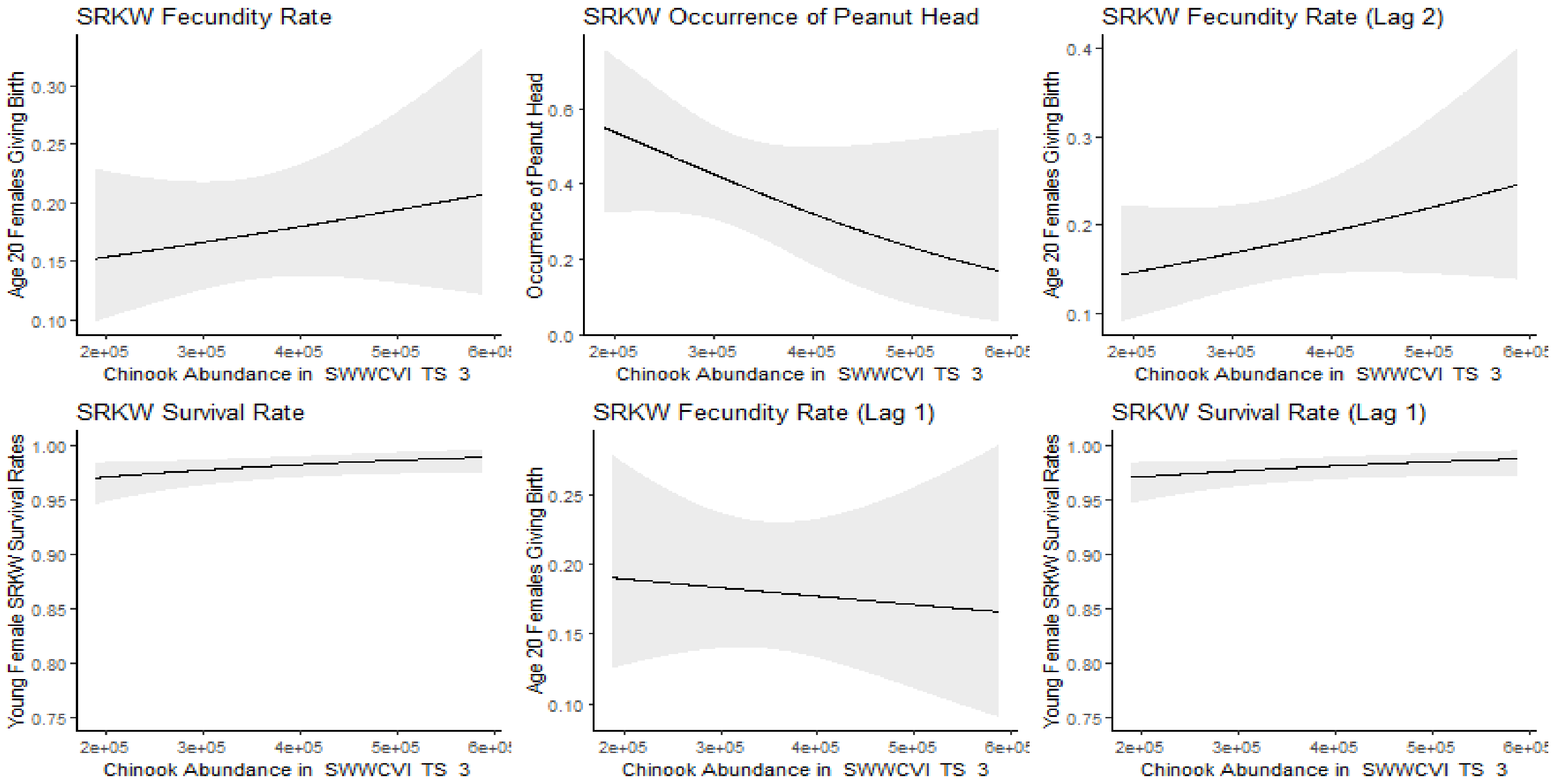
Appendix C Figure 13. South West / West Coast of Vancouver Island aggregate Timestep 1



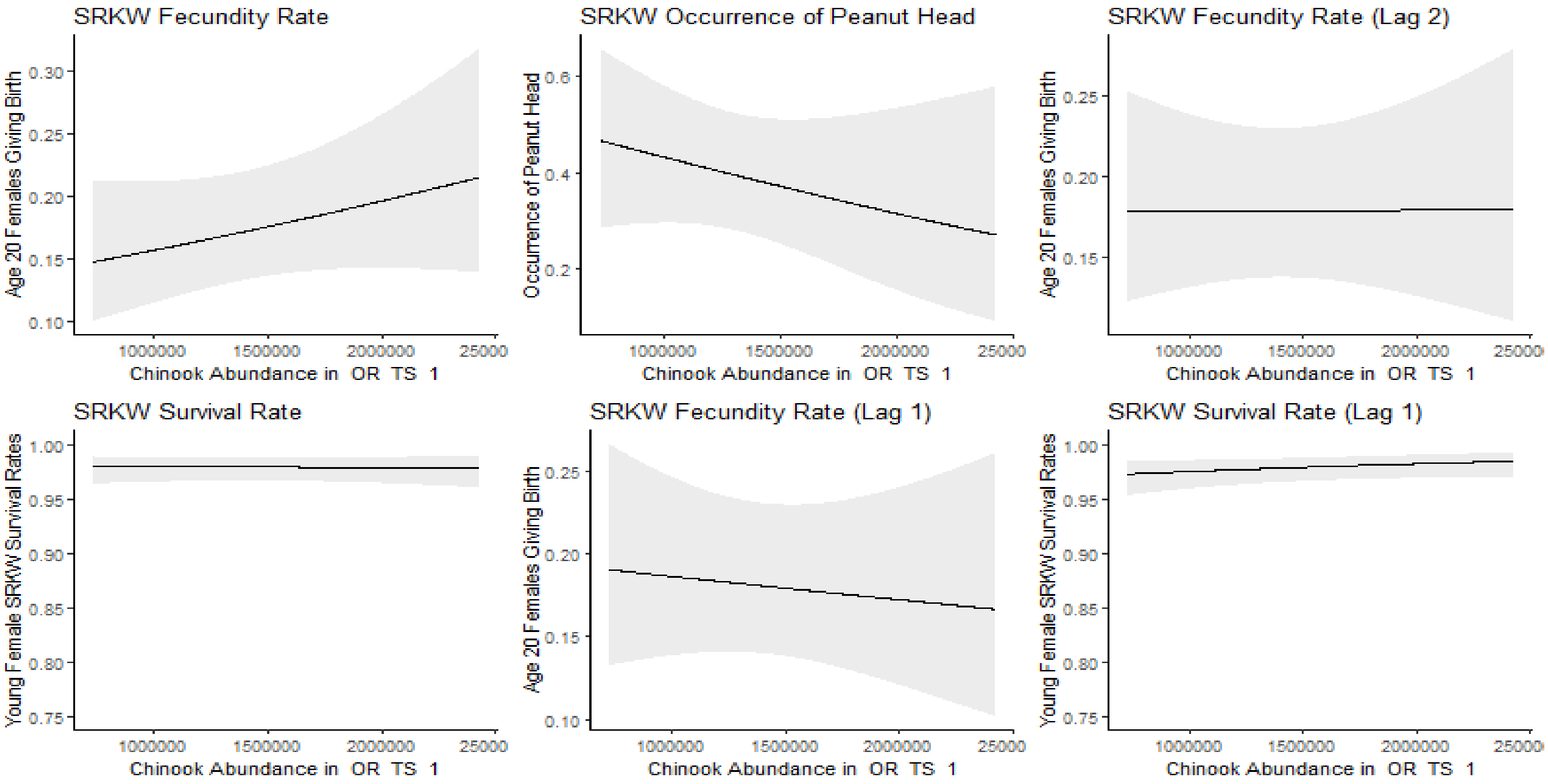
Appendix C Figure 14. South West / West Coast of Vancouver Island aggregate Timestep 2



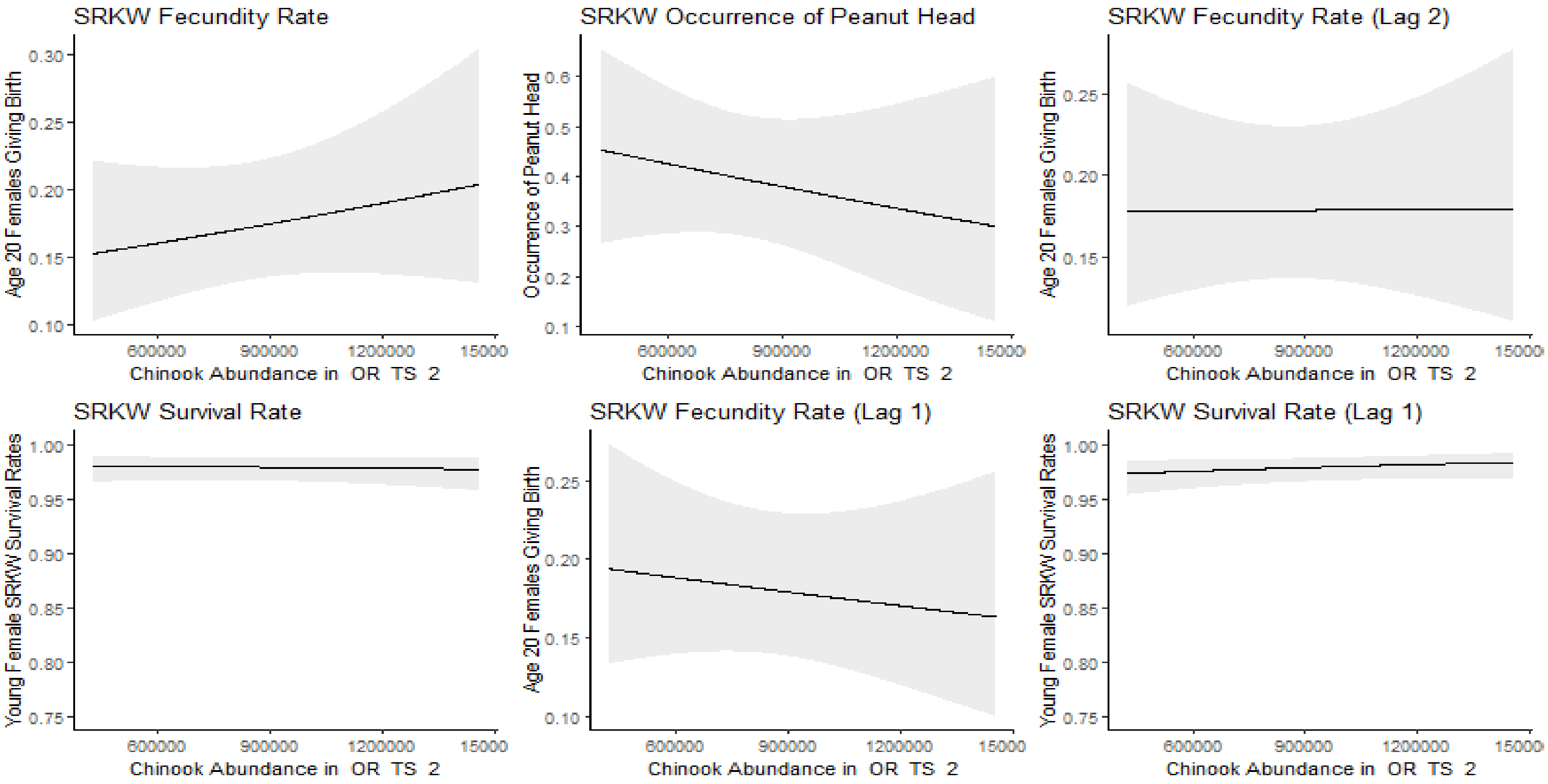
Appendix C Figure 15. South West / West Coast of Vancouver Island aggregate Timestep 3



Appendix C Figure 16. Oregon Coast Timestep 1



Appendix C Figure 17. Oregon Coast Timestep 2


SRKW Fecundity Rate

SRKW Occurrence of Peanut Head

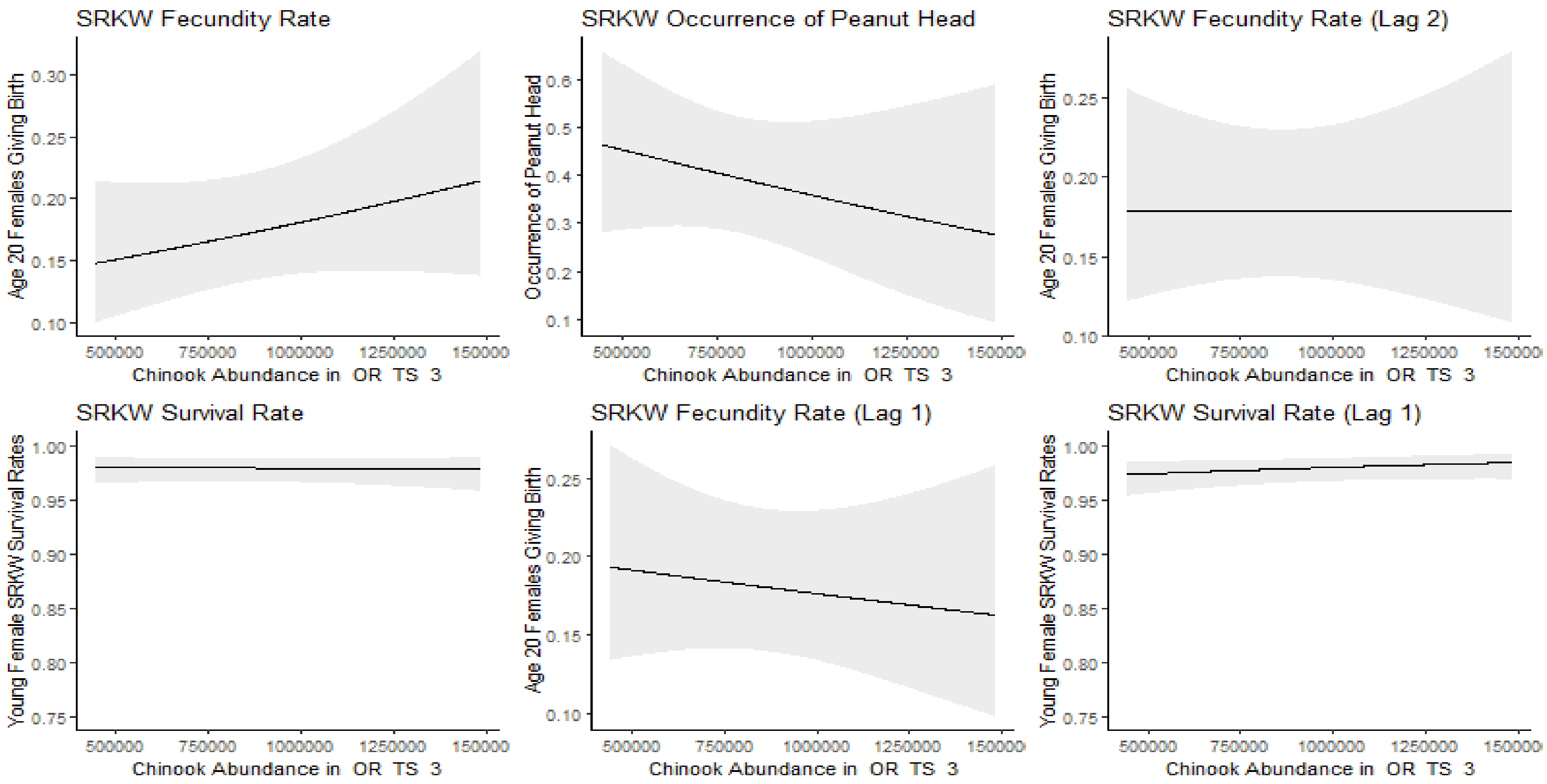
SRKW Fecundity Rate (Lag 2)

SRKW Survival Rate

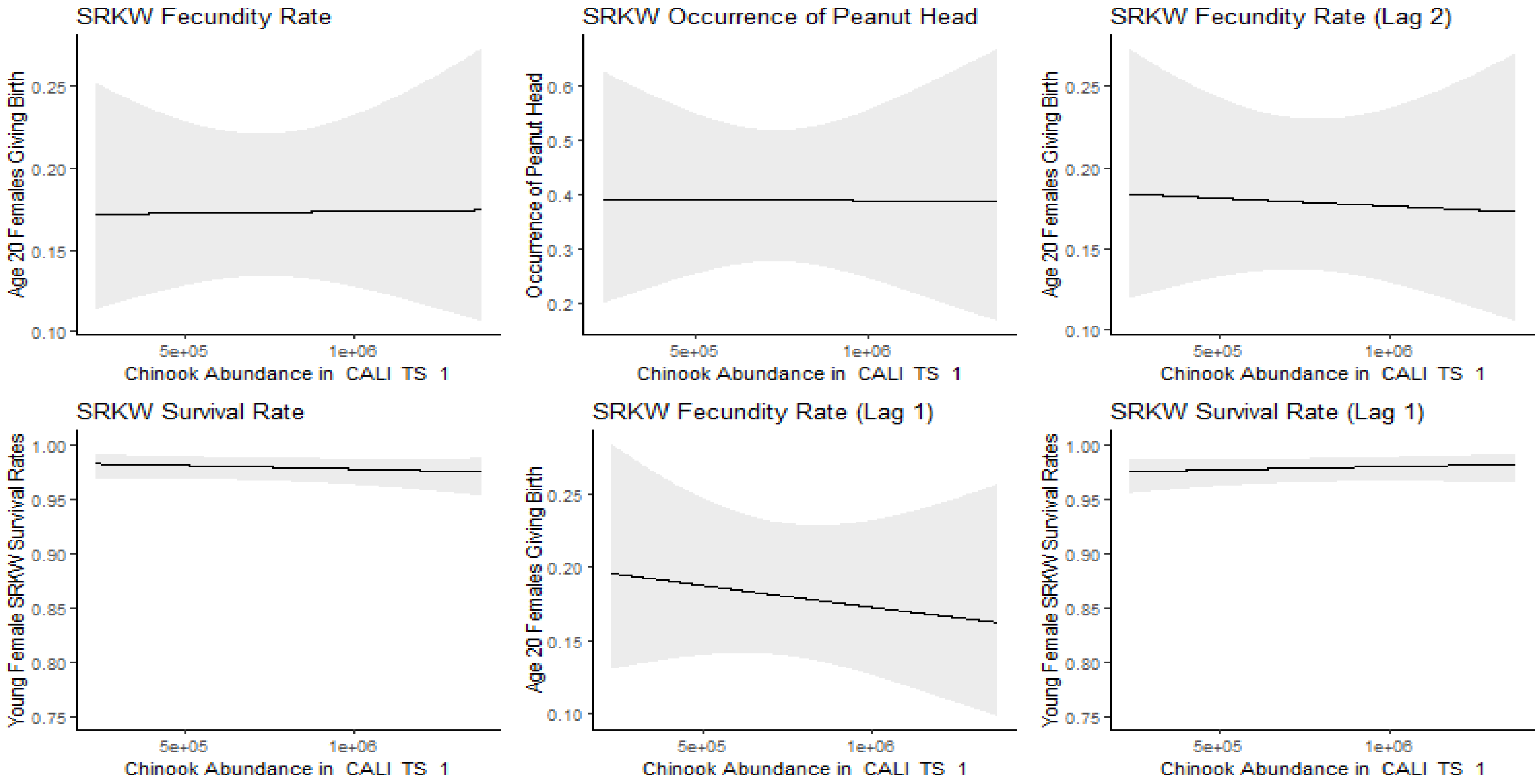
SRKW Fecundity Rate (Lag 1)

SRKW Survival Rate (Lag 1)

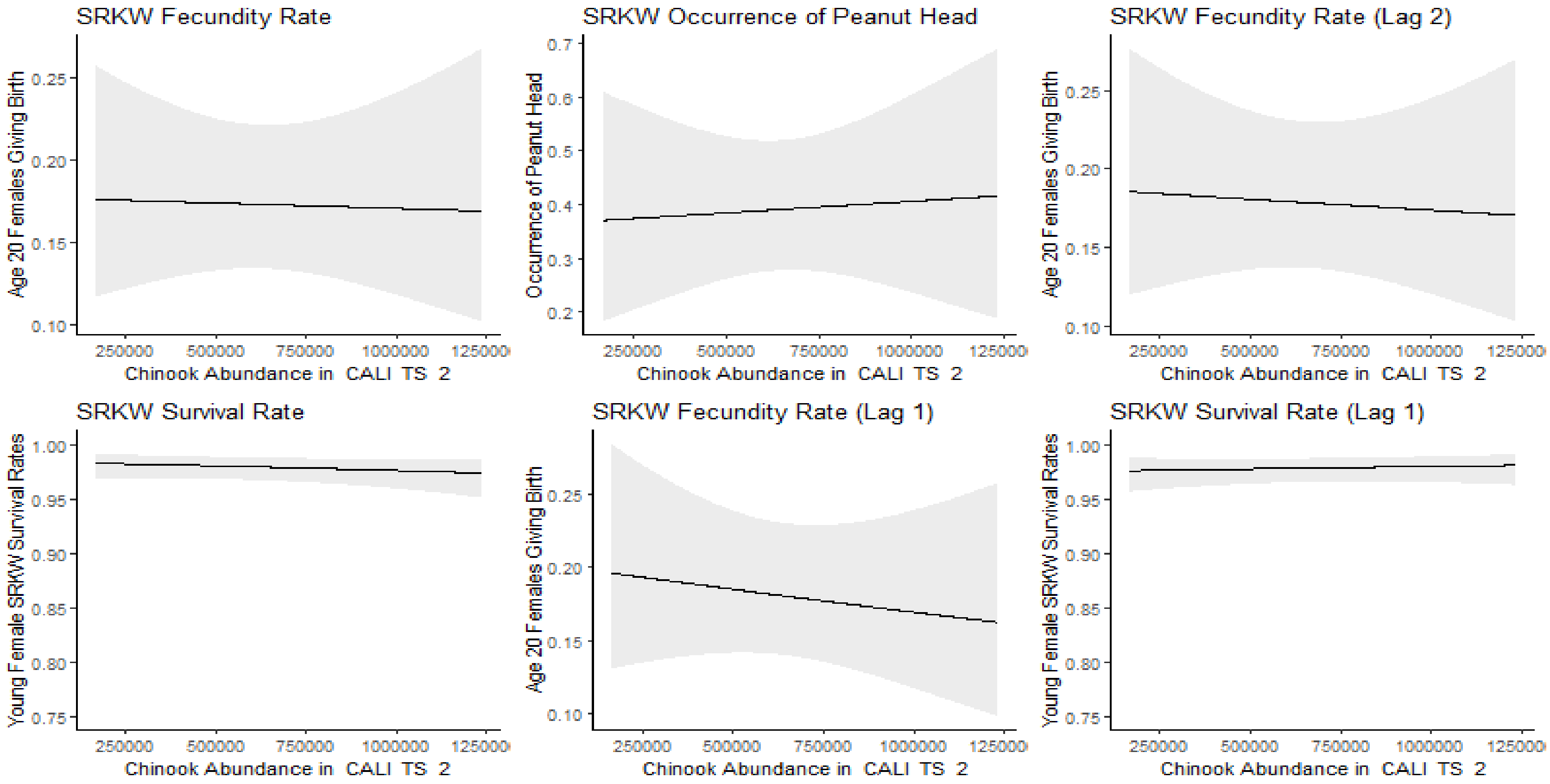
Appendix C Figure 18. Oregon Coast Timestep 3



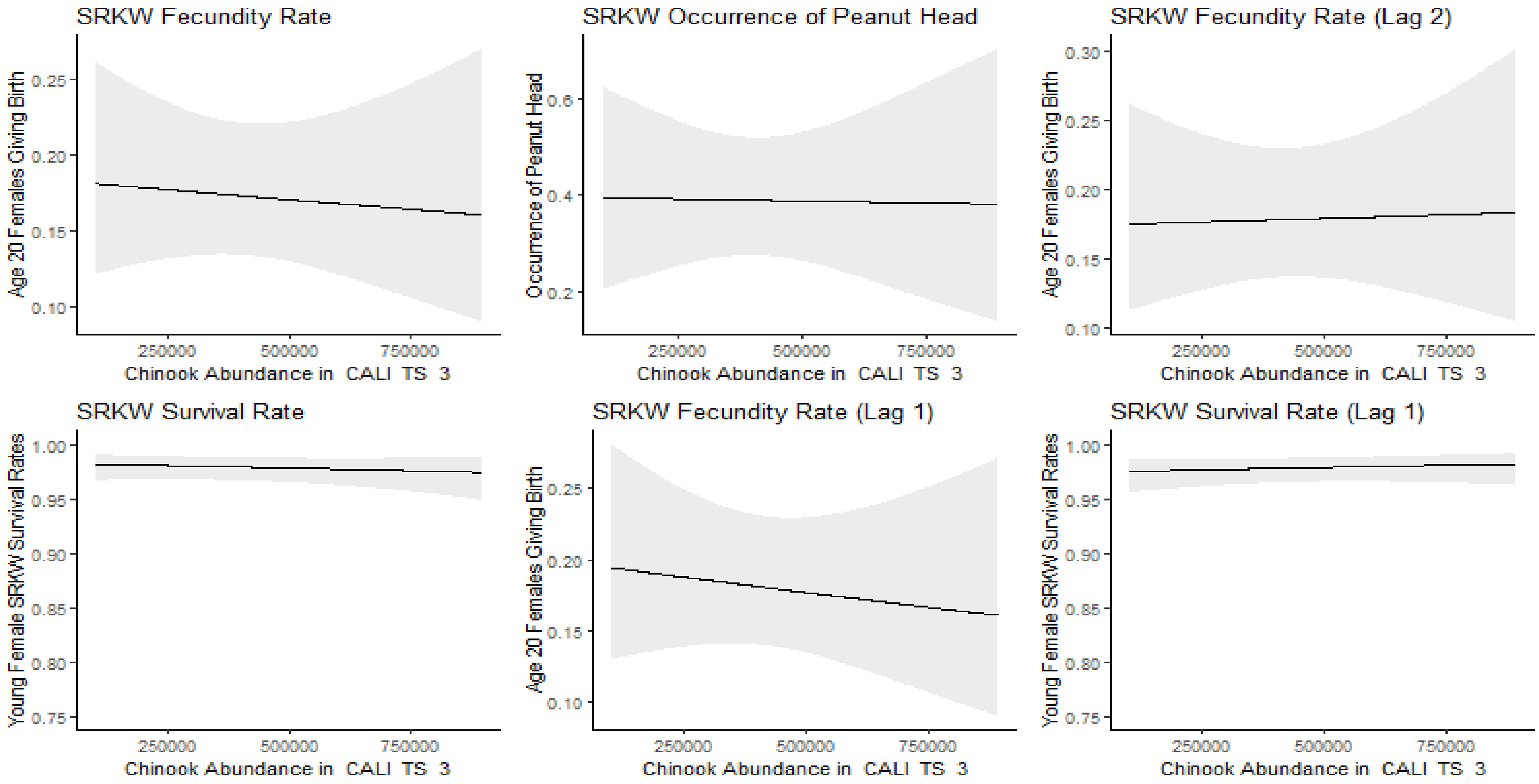
Appendix C Figure 19. California Coast Timestep 1



Appendix C Figure 20. California Coast Timestep 2



Appendix C Figure 21. California Coast Timestep 3



SRKW Fecundity Rate

SRKW Occurrence of Peanut Head

SRKW Fecundity Rate (Lag 2)

SRKW Survival Rate

SRKW Fecundity Rate (Lag 1)

SRKW Survival Rate (Lag 1)

APPENDIX D

Clustering Analysis

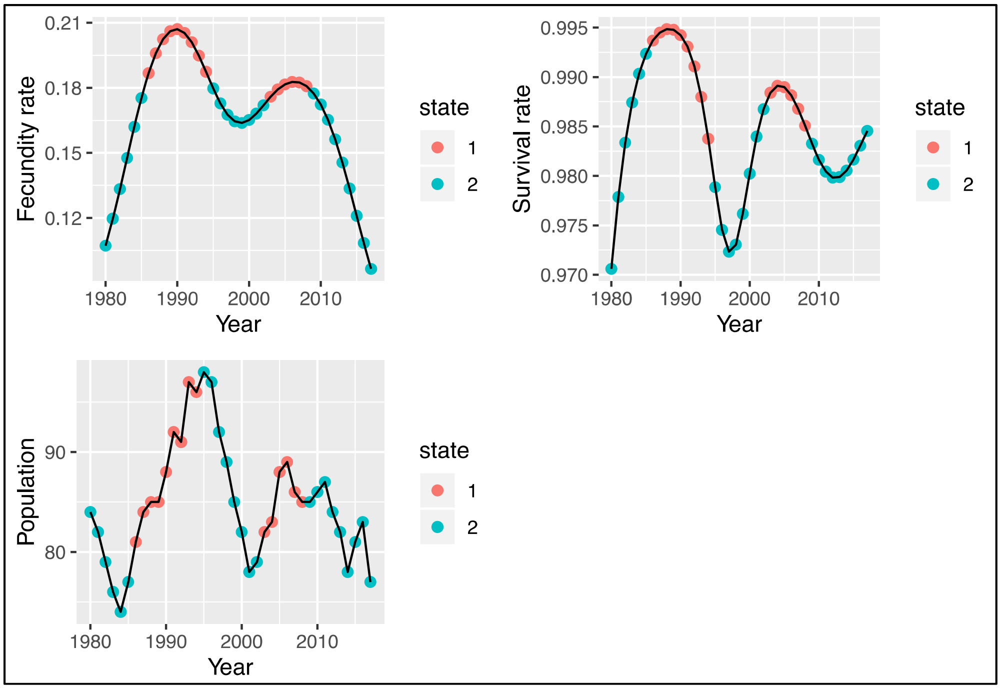
SRKW population trends were also considered as an assessment metric for clustering analyses attempting to identify years of high versus low risk to SRKW, represented as a binomial variable with 1 corresponding to population increases and 0 corresponding to no population increases (including decreases or no growth). Periods of population increase and decrease/no growth were estimated by fitting a GAM (total SRKW population ~ year), with inflection points in the GAM representing changes in the direction of the population trend. Unlike fecundity, survival, or the occurrence of peanut head, relationships between Chinook salmon abundance and SRKW population trends were not examined in isolation because population increase/decrease is a simple function of births and deaths, which can be modeled separately while also accounting for the effects of age/sex structure and deaths that are clearly not food-related.

In the workgroup's first attempt at cluster analysis (July 2019), we used partitioning around medoids (PAM) to explore possible associations between Chinook salmon abundance and the SRKW population metrics (fecundity, survival, occurrence of whales with peanut head condition, indicators of SRKW population trends). This analysis grouped together years based on annual summary modeled (estimated) values for selected demographic variables (details on statistical smoothing available at https://www.fisheries.noaa.gov/webdam/download/94054344), optimizing the degree of association between the values of variables examined. For example, one group of years may be associated with high values for fecundity, survival, and SRKW population trends, but low values for the occurrence of peanut head syndrome. Thus, these years may be assigned to a cluster of years that represent above average demographic rates or population performance. Local Chinook salmon abundance can also be considered in defining clusters, or related to assigned clusters post-hoc. For the cluster analysis, we considered between two and four groups of years to examine associations. These clustering analyses are documented in the link referenced above, but the workgroup could not fully interpret the analyses with more than two groups, and had concerns about how the clustering algorithm treated variables with very different ranges (*e.g.*, annual survival indices varied from 0.969 to 0.997 while coastwide Chinook abundance varied by millions of fish), scales (*e.g.*, the maximum fecundity rate was 0.231), and distributional forms (e.g. some metrics, like demographic rates were continuous variables, while indicators of population growth were binary).

As a second approach, we refined the cluster analysis in October 2019, with several important changes. First, the working group voted to not include the occurrence of peanut head syndrome as an indicator. While the occurrence of peanut head syndrome has the potential to include new information not accounted for by other metrics, concerns were that it has the potential to be redundant with survival estimates and the quality of the data before 1994 is unknown. As a second update, the group limited the analysis to only including two clusters, rather than two to four. Third, the group switched the modeling approach away from the partitioning around medoids algorithm to an approach that better allowed for mixed data types. Specifically, we fit a two state mixture model with varying data types (fecundity and survival modeled with Gaussian distributions, population growth as a binomial response). We used the expectation – maximization (EM) algorithm to find the maximum likelihood solution. Given the known issues of these approaches getting trapped in false maxima, we initialized estimation from 100 independent starting values. We used the R package depmixS4 (Visser and Speekenbrink 2010) to perform all estimation. After clusters were assigned, we examined the distribution of Chinook abundances by cluster to examine the utility in using clusters to also identify better and worse periods of salmon abundance (repeating this analysis by season and area).

Results from clustering analysis applied to SRKW demographic indices and specifying 2 clusters identified several periods of relatively higher and lower risk for SRKW (because 2 clusters were specified *a priori*, the model only identified these two states). Because of the known label-switching problem with clustering approaches, we examined the relationship between the cluster centers for each variable and corresponding state. For these results, we are assigning state 2 to represent periods of higher risk because it is associated with lower survival, lower birth rates, and not increasing population size (Appendix D Figure.1). Periods of low risk (state 1) that the model identified were 1986-1994, and 2003-2008. It should be noted that the cluster assignments may not appear to capture trends and variability perfectly (*e.g.* the pulse of births in 2014-2015 does not appear to be captured by the trend in fecundity rates, and are assigned to state 2). These discrepancies are caused by the inputs to the clustering algorithm being time series of predicted values estimated from a generalized additive model (GAM). If other approaches were used instead (*e.g.* year effects treated as fixed or random effects), we would expect different cluster assignments.

Another source of discrepancy is that the cluster assignment represents the state that on average explains trends in population size and variability in demographic rates. There may be periods where one or more of these indicators is not in agreement with the other two. Because cluster analyses may be sensitive to the choice of initial values, we repeated the clustering algorithm from 100 starting values, and used AIC to select the most parsimonious clustering across replicates (many of these resulted in identical cluster assignments, however several iterations appeared to converge on solutions with lower likelihoods). Finally, we examined the utility of linking the assigned states (lower, higher risk) to periods of below or above average Chinook salmon abundance. We examined the average estimated abundance (aggregated across stocks) by time period and area and found little relationship; in other words the 'lower risk' periods 1986-1994 and 2003-2008 corresponded to years with both below and above average Chinook abundance.



Appendix D Figure.1. Clustering analysis for a two *a priori* state evaluation to determine if periods of relatively higher and lower risk for SRKW demographics were present between 1980 and 2016.

Appendix D References

Ingmar Visser, Maarten Speekenbrink (2010). depmixS4: An R Package for Hidden Markov Models. Journal of Statistical Software, 36(7), 1-21. URL: http://www.jstatsoft.org/v36/i07/.

APPENDIX E

Fishery Reductions

The following tables capture starting abundances during time step one, meaning starting abundances in the October-April time step for "zero PFMC" fishing runs. They also capture the subsequent annual fishery abundance reduction, which represents the difference between post-fishing abundances for a "zero PFMC fishing scenario" and FRAM validation run in time step 3 (e.g., total mortalities resulting from fisheries on the abundance across the entire management year). These numbers are reported here in the stratifications used in the methodology describe in Section 5.1, with the resulting percent reductions calculated from the annual estimated reduction in fishing mortality.

Appendix E Table 1. Coastwide (EEZ).

Year	Starting abundance in Oct-April Timestep	PFMC Fishery Abundance Reduction	Percent Reduction
1992	2,171,837	401,746	-18.5%
1993	2,852,633	603,265	-21.1%
1994	2,308,645	530,954	-23.0%
1995	4,052,786	1,221,886	-30.1%
1996	3,308,575	761,013	-23.0%
1997	3,339,343	878,994	-26.3%
1998	2,497,949	478,280	-19.1%
1999	2,649,865	433,266	-16.4%
2000	3,449,629	678,250	-19.7%
2001	4,766,907	571,350	-12.0%
2002	5,892,136	882,176	-15.0%
2003	5,691,225	1,005,578	-17.7%
2004	5,076,303	1,312,870	-25.9%
2005	3,843,132	713,720	-18.6%
2006	2,805,717	447,109	-15.9%
2007	2,123,292	257,870	-12.1%
2008	2,217,428	63,551	-2.9%
2009	2,244,311	19,128	-0.9%
2010	3,853,767	109,086	-2.8%
2011	3,232,247	151,274	-4.7%
2012	4,391,730	447,756	-10.2%
2013	5,995,923	645,345	-10.8%
2014	4,637,261	555,691	-12.0%
2015	4,847,834	313,344	-6.5%
2016	2,803,180	168,506	-6.0%
Time series average	3,642,146	546,080	-15.0%
Recent 10 year average	3,634,697	273,155	-7.5%

Appendix E Table 2. North of Falcon

Year	Starting abundance in Oct-April Timestep	PFMC Fishery Abundance Reduction	Percent Reduction
1992	1,030,292	55,558	-5.4%
1993	1,081,600	55,209	-5.1%
1994	814,339	31,079	-3.8%
1995	1,020,578	77,287	-7.6%
1996	1,034,548	54,813	-5.3%
1997	1,145,839	60,807	-5.3%
1998	861,578	41,534	-4.8%
1999	1,039,156	37,638	-3.6%
2000	1,035,805	47,961	-4.6%
2001	1,892,271	80,305	-4.2%
2002	2,095,142	124,029	-5.9%
2003	1,921,000	135,608	-7.1%
2004	1,935,286	140,922	-7.3%
2005	1,458,648	102,071	-7.0%
2006	1,287,054	44,425	-3.5%
2007	946,614	29,104	-3.1%
2008	1,232,760	17,224	-1.4%
2009	1,050,337	12,033	-1.1%
2010	1,903,168	49,959	-2.6%
2011	1,503,600	39,165	-2.6%
2012	1,539,986	65,879	-4.3%
2013	2,422,663	88,413	-3.6%
2014	1,940,237	113,884	-5.9%
2015	2,247,447	88,497	-3.9%
2016	1,430,091	35,438	-2.5%
Time series average	1,434,802	65,154	-4.5%
Recent 10 year average	1,621,690	53,960	-3.3%

Appendix E Table 3. Salish Sea

Year	Starting abundance in Oct-April Timestep	PFMC Fishery Abundance Reduction	Percent Reduction
1992	618,530	13,878	-2.2%
1993	598,571	12,460	-2.1%
1994	433,464	2,248	-0.5%
1995	499,983	5,067	-1.0%
1996	512,248	5,223	-1.0%
1997	686,651	8,642	-1.3%
1998	502,538	7,795	-1.6%
1999	639,218	11,287	-1.8%
2000	435,168	5,849	-1.3%
2001	709,973	13,856	-2.0%
2002	693,863	18,863	-2.7%
2003	680,928	21,267	-3.1%
2004	670,488	20,587	-3.1%
2005	602,904	17,938	-3.0%
2006	677,486	11,139	-1.6%
2007	546,750	8,920	-1.6%
2008	601,332	6,658	-1.1%
2009	442,066	4,093	-0.9%
2010	826,614	14,972	-1.8%
2011	609,152	10,778	-1.8%
2012	523,264	15,819	-3.0%
2013	742,819	16,094	-2.2%
2014	637,945	19,513	-3.1%
2015	643,377	15,442	-2.4%
2016	569,727	8,001	-1.4%
Time series average	604,202	11,856	-2.0%
Recent 10 year average	614,305	12,029	-2.0%

Appendix E Table 4. Southwest West Coast Vancouver Island

Year	Starting abundance in Oct-April Timestep	PFMC Fishery Abundance Reduction	Percent Reduction
1992	537,078	13,346	-2.5%
1993	527,786	12,820	-2.4%
1994	416,787	5,879	-1.4%
1995	489,750	14,255	-2.9%
1996	516,750	10,870	-2.1%
1997	519,479	12,432	-2.4%
1998	428,508	9,546	-2.2%
1999	512,225	9,493	-1.9%
2000	416,504	9,502	-2.3%
2001	764,131	17,705	-2.3%
2002	902,558	26,648	-3.0%
2003	873,012	30,688	-3.5%
2004	906,749	30,910	-3.4%
2005	723,078	23,056	-3.2%
2006	648,572	11,010	-1.7%
2007	483,504	8,008	-1.7%
2008	605,867	4,775	-0.8%
2009	509,038	3,277	-0.6%
2010	875,000	12,176	-1.4%
2011	725,120	9,645	-1.3%
2012	724,280	16,689	-2.3%
2013	1,275,291	19,141	-1.5%
2014	942,889	24,136	-2.6%
2015	1,118,100	19,790	-1.8%
2016	723,352	8,642	-1.2%
Time series average	686,616	14,578	-2.1%
Recent 10 year average	798,244	12,628	-1.6%

Appendix E Table 5. Oregon coast (Cape Falcon south to OR/CA border) Horse Mountain, California.

Year	Starting abundance in Oct-April Timestep	PFMC Fishery Abundance Reduction	Percent Reduction
1992	764,075	130,732	-17.1%
1993	1,130,577	215,500	-19.1%
1994	904,476	170,088	-18.8%
1995	1,779,890	392,933	-22.1%
1996	1,399,384	283,938	-20.3%
1997	1,247,445	276,918	-22.2%
1998	981,937	150,366	-15.3%
1999	915,724	136,653	-14.9%
2000	1,438,901	235,841	-16.4%
2001	1,829,503	203,868	-11.1%
2002	2,379,489	302,207	-12.7%
2003	2,455,548	418,469	-17.0%
2004	1,998,112	529,780	-26.5%
2005	1,466,795	246,295	-16.8%
2006	954,271	161,094	-16.9%
2007	791,496	103,614	-13.1%
2008	743,605	35,520	-4.8%
2009	920,183	5,892	-0.6%
2010	1,495,958	38,923	-2.6%
2011	1,268,829	59,678	-4.7%
2012	1,934,006	179,238	-9.3%
2013	2,422,163	254,837	-10.5%
2014	1,878,195	211,563	-11.3%
2015	2,002,224	121,791	-6.1%
2016	1,009,659	60,417	-6.0%
Time series average	1,444,498	197,046	-13.6%
Recent 10 year average	1,446,632	107,147	-7.4%

Appendix E Table 6. California coast, south of Horse Mountain

Year	Starting abundance in Oct-April Timestep	PFMC Fishery Abundance Reduction	Percent Reduction
1992	377,470	215,456	-57.1%
1993	640,455	332,556	-51.9%
1994	589,829	329,787	-55.9%
1995	1,252,318	751,666	-60.0%
1996	874,644	422,262	-48.3%
1997	946,059	541,269	-57.2%
1998	654,434	286,380	-43.8%
1999	694,984	258,976	-37.3%
2000	974,923	394,447	-40.5%
2001	1,045,132	287,177	-27.5%
2002	1,417,505	455,941	-32.2%
2003	1,314,678	451,501	-34.3%
2004	1,142,905	642,169	-56.2%
2005	917,689	365,353	-39.8%
2006	564,393	241,590	-42.8%
2007	385,182	125,151	-32.5%
2008	241,063	10,807	-4.5%
2009	273,791	1,203	-0.4%
2010	454,641	20,203	-4.4%
2011	459,818	52,431	-11.4%
2012	917,738	202,640	-22.1%
2013	1,151,097	302,095	-26.2%
2014	818,829	230,244	-28.1%
2015	598,163	103,057	-17.2%
2016	363,430	72,651	-20.0%
Time series average	762,847	283,881	-37.2%
Recent 10 year average	566,375	112,048	-19.8%