Submission to NMFS Five-Year Status Review of Southern Resident Killer Whales

Submitted by

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To Whom It May Concern,

We appreciate the opportunity to provide NMFS with information and recommendations for its 2021 Five-year Review of the Status of Southern Resident Killer Whales (SRKW) distinct Population Segment (DPS). Herein, we provide several documents pertaining to recent population viability analyses (PVAs) of the SRKW DPS. All demonstrate that the probability of quasi-extinction to 30 animals or less, or to only one sex remaining exceed 20% (0.20) within fewer than 100 years. This is a dire situation for the DPS and indicates that NMFS' 2008 SRKW Recovery Plan and related management actions by NMFS are failing to secure the requisite high probability of survival required to prevent extinction, much less initiate the recovery of the DPS.

We also recommend that the Biological Review Team (BRT) conducting the review evaluate the PVAs described in Lacy et al. 2017 and Murray et al. 2021 to inform both management actions and identify a PVA modeling approach for NMFS. It is critical that the public and scientific community have confidence that NFMS' approach to estimating SRKW extinction risk is scientifically robust and credible. It is further critical that NMFS recognize that the current status of the DPS, based on the evidence of the PVAs of Lacy et al. and Murray et al., is dire and requires strong precautionary actions that address the primarily limiting factors to survival and recovery over which NMFS and other federal and state agencies have the most direct control.

Specifically, as we discuss in further detail below, marine Chinook harvest requires far greater consideration to mixed stock catch removals, Chinook abundance within SRKW foraging grounds, population structure including size and age composition, and lastly precautionary modelling and management approaches that reflect a current understanding of rapidly changing marine environments that are difficult to predict and dictate far greater consideration to ecosystem function than are currently in place.

Evidence from the last decade shows a poor ability to adequately predict Chinook abundance and allowable harvest limits in mixed stock west coast fisheries in Canada and the US. This results in fishing above catch ceilings set by the PST[1] and a lower abundance of Chinook salmon in SRKW critical habitat and feeding grounds. As NMFS has identified, low Chinook salmon abundance has been associated with low killer whale fecundity and survival (Hanson et al. 2021). Chinook removal above catch ceilings further reduces prey to SRKWs from runs known generally to be important in their diets, such as those that return to Puget Sound, Georgia Strait, Canada's Fraser River (summer and fall runs) and the Columbia River (spring, summer and fall runs), as well as other coastal stocks returning to Washington, Oregon and California.

In addition to imprecise and inaccurate aggregate abundance forecasts, there are considerable uncertainties regarding the appropriate Chinook stock aggregate and levels of pre-season abundance that correlate to positive SRKW fecundity and survival rates. Additionally, a critically important feature of an appropriate Chinook stock aggregate is the age- and size-composition.

Preferred prey

It is well-known that SRKW are highly specialized predators that feed primarily on Chinook salmon. What seems less appreciated is the importance of Chinook age and size that constitutes their preferred prey. Ford and Ellis (2005) found resident killer whales preferentially select for Chinook that are four and five years or older with corresponding body sizes of greater than 740

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[1] AABM fisheries managed under the PST occur in Alaska and British Columbia. Alaska has exceeded its catch ceilings (by 5% or greater) more than 50% the time since 2010 (range is from 6.48% (2010: ~15,000 fish) to 31.42% (2020: ~64,000 fish). By comparison, BC (combined NBC & WCVI) exceeded its catch ceiling once (8.36% in 2016, representing ~13,000 fish).

mm fork length (29 inches) and body masses of at least 8-13 kg (17 lbs or more). Analyses by NMFS's Dr. Eric Ward shows strong selectivity (preference) by SRKW for Chinook five years (> 800 mm FL) and older. NMFS has so far failed to account for age and size criteria as a fundamental aspect of preferred prey. Historically, these Chinook ages and sizes would have comprised the majority of Chinook salmon stocks with which SRKW co-evolved. As age at maturity and size at age of Chinook salmon has declined over the last century, half century, and in recent decades, the proportional abundance of the larger, older Chinook (i.e. preferred prey) has also declined. This decline is in addition to observed declines of aggregated Chinook salmon abundance regionally and internationally.

Age overfishing

In addition to lowering the overall abundance of primary prey within critical habitat, mixed stock marine Chinook fisheries also harvest immature Chinook, as these fisheries are conducted on Chinook salmon rearing grounds. The harvest of immature Chinook skews the age composition of Chinook populations towards younger and hence smaller individuals. A reduction in the size and number of older females has adverse implications for Chinook productivity (as fecundity is related to the length of female Chinook salmon), as well as for Chinook recovery and resilience, (since large female Chinook have advantages on spawning grounds that improve egg deposition and survival). So in addition to fishery removals that directly lower Chinook abundance, marine Chinook fisheries can skew the age structure toward younger and smaller fish, and a lower abundance of larger older preferred prey for SRKWs.

NMFS needs to determine the fisheries management measures required to ensure the DPS has a high probability of securing regular annual access to the minimal numbers of large (>740 mm FL) Chinook conservatively estimated to be necessary to secure the survival of the current SRKW population and to provide modest annual increase in population numbers. The Status Review should determine the extent to which current Chinook salmon harvest management measures have failed to secure these levels of abundance of Chinook with the requisite age and size composition.

Preserving reproductive potential

Premature deaths of adult and juvenile SRKWs (i.e. mortality rates up to 2-3 times greater than expected; DFO 2017) have been occurring within all three pods for almost two decades. These events underscore the precarious nature of this population whose individuals are generally failing to successfully feed, mature and reproduce, or meet normal life expectancies. In this small and declining population, reproductive potential is eroded with every death regardless of sex or age (due to the cultural importance of post-reproductive females and the concern for inbreeding from too few suitable mature males).

A critical attribute of the current SRKW DPS demographic is the low birth rate. Even recent minor improvements to birth rates in 2019 – 2021 are still below the number of annual births expected for a healthy resident killer whale population, as is generally the case for Southern Alaska Resident Killer Whales (SARKW) (Murray et al. 2021, Table 1). Additionally, the challenge for more than a decade has been keeping young calves (and fetuses in pregnant females) alive to become breeding adults. The cause is largely attributed to the poor nutritional condition of the mothers (Wasser et al. 2017).

Compared to a healthy resident killer whale population, even the recent minor uptick in SRKW births is low. Murray et al. 2021 (attached) use demographic data from the Southern Alaska Resident Killer Whale (SARKW) population as a reference for a healthy resident killer whale population against which to compare the demographic performance of the SRKW population. Their Table 1 lists the following parameters for the mean annual birthrate of mature females in the SARKW population:

- Females ages 10 to 30: mean = 0.233, standard deviation = 0.118,
- Females ages 31 to 50: mean = 0.154, standard deviation = 0.118.

As of 2020, there are 18 females between the ages of 10 and 30 in the SRKW population, and 16 females between the ages of 31 to 50. The mean annual birth rates are Bernoulli rates (probabilities of an individual mature female having a birth in year x, and, hence, the number of annual births of a group of similar individuals each of which has the same Bernoulli probability of birth) will be distributed as a binomial with parameters n (number of females) and p (probability of in individual female giving birth in year x).

The Beta Distribution is the parametric distribution for a binomial rate parameter. Accordingly, to account for the variability in the mean birth rates of the SARKW population given in Table 1 of Murray et al. 2021, we simulated (in Matlab) 50000 binomial trials of annual births with parameters

- n1(females ages 10 to 30) = 18, and p = p-female1; and
- n2 (females ages 31 to 50) = 16, p-female2,

where p-female1 is drawn randomly for each trial from a Beta distribution with alpha parameter = 2.76 and beta parameter = 9.08, which yields a mean p of 0.233 with standard deviation of 0.188. P-female2 is drawn randomly from a Beta distribution with alpha = 1.287 and beta = 7.071, which yields a mean p of 0.154 with standard deviation of 0.118. The resulting distribution of annual births from 18 females ages 10 to 30 and from 16 females ages 31 to 50 will each be distributed as a Beta-Binomial Distribution. This provides a robust estimate of the probability distribution of expected annual births to females1 and females 2 and to their sum, the total number of annual births expected from all 34 mature females. Summary results for all 34 mature females are listed in the table below;

Table 1. Probabilities of numbers of annual births for a healthy resident killer whale population *f* with a total of 34 mature females between the ages of 10 and 50. ‘P(Interval)’ is the probability of the annual number of births falling within the number of births in the ‘Interval’ column.

#Births	P(#Births)	Cumulative P	Interval	P(Interval)
0	0.06	0.06	2 to 4	0.20
1	0.11	0.17	2 to 5	0.23
2	0.06	0.22	3 to 5	0.15
3	0.08	0.30	3 to 6	0.29
4	0.12	0.43	3 to 7	0.29
5	0.02	0.45	3 to 8	0.41
6	0.14	0.59	4 to 6	0.16
7	0.01	0.60	4 to 7	0.17
8	0.11	0.71	4 to 8	0.28
9	0.02	0.73	5 to 9	0.28
10	0.08	0.81	5 to 10	0.36

Significantly, the probability that the mean number of annual births is greater than 3, is greater than 70% (1 – 0.30 = 0.70 (70%)), the probability that the mean number is greater than 4 or 5, is greater than 55% (1-0.43 = 0.57 (57%)); 1 – 0.45 = 0.55 (55%), respectively). The probability that the mean number of births is greater than 6 or 7, is greater than 40% (1 - 0.59 = 0.41 (41%)); 1 – 0.60 = 0.40 (40%), respectively). The maximum number of births in the SRKW population over the past decade has never exceeded 3, and most frequently has been between 0 and 2. This is clearly abnormally low for a resident killer whale population. Addressing this needs to be reflected in the new Five Year Review.

Further, as we show below, the recent (2019 to 2021) uptick in births may be the result in large part of recent reductions in Chinook harvest in British Columbia due to a combination of management actions by DFO (2018- 2020) and reduced fishing pressure from Covid-19 measures in effect through 2020. The BRT needs to carefully evaluate this matter, as it likely significantly affects the understanding of the potential for further reductions in coastwide Chinook harvests to achieve significant near-term benefits to the DPS.

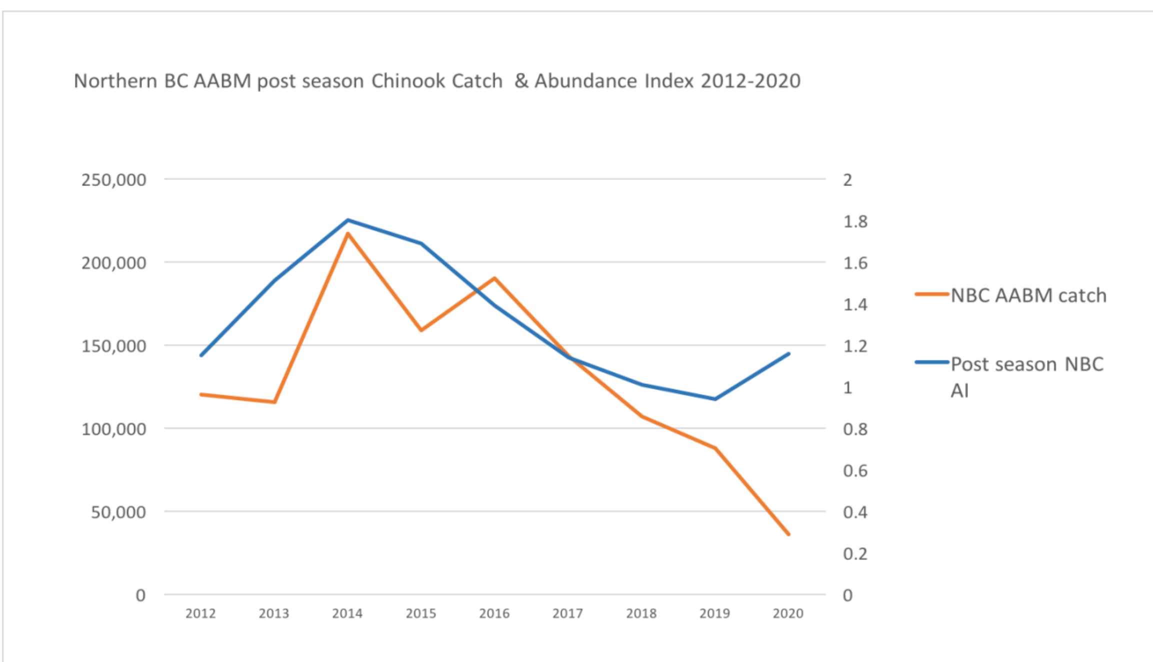

Figure 1. Post season Abundance Index (AI, blue) as determined by the PSC's Chinook Technical Committee for Northern BC and the observed Chinook catch (orange) in Northern BC

from 2012- 2020. Northern BC is one of three west coast fisheries managed under the PST's Aggregate Abundance Based Management (AABM). Catches are affected by the forecasted pre-season abundance and its corresponding catch ceilings (not shown), and then evaluated against observed abundance post season. The index is a relative indicator of Chinook abundance. The highest index for NBC since 2012 was 1.8 in 2014. Greater deviation in catches from the AI since 2018 are due to reduced fishing pressure from domestic (BC) Chinook management measures and response measures (such as travel restrictions) in 2020 for Covid-19.

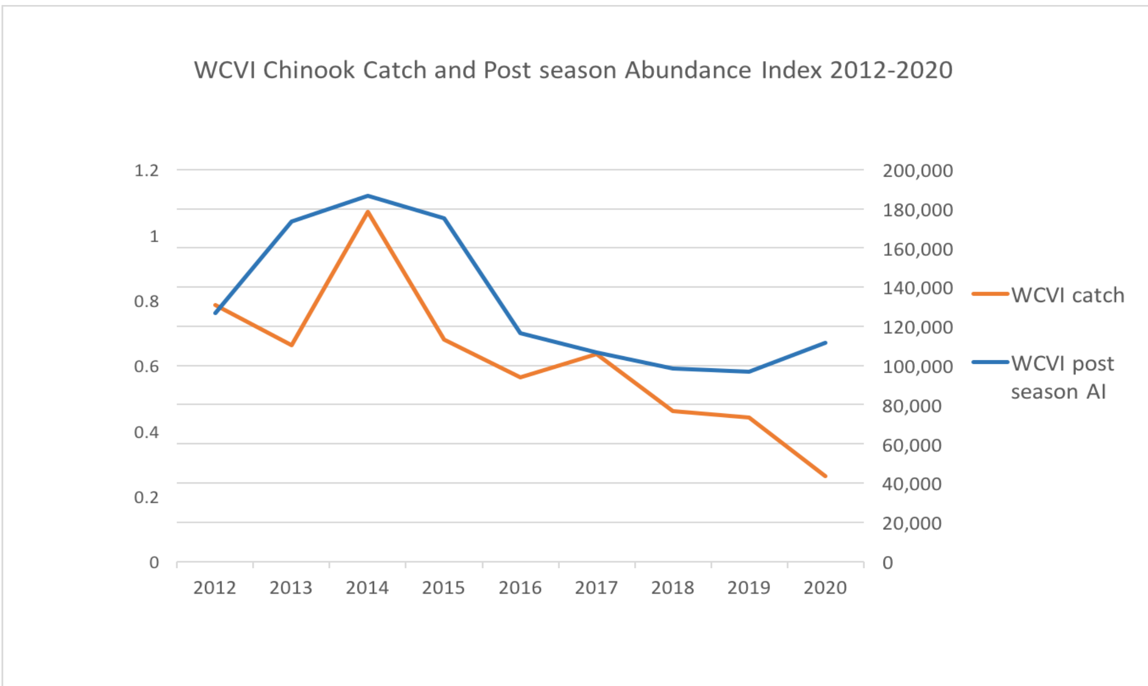

Figure 2. Post season Abundance Index (AI, blue) as determined by the PSC's Chinook Technical Committee for the West Coast of Vancouver Island and the observed Chinook catch (orange) in WCVI from 2012- 2020. WCVI is one of three west coast fisheries managed under the PST's Aggregate Abundance Based Management (AABM). Catches are affected by the forecasted pre-season abundance and its corresponding catch ceilings (not shown), and then evaluated against observed abundance post season. The index is a relative indicator of Chinook abundance. The highest index for WCVI since 2012 was 1.12 in 2014. Greater deviation in catches from the AI since 2018 are due to reduced fishing pressure from domestic Chinook management measures in BC and response measures (such as travel restrictions) in 2020 for Covid-19.

Table 2. Post Season Abundance Index (AI) for the three AABM fishery areas (SEAK, NBC and WCVI) and the Total Allowable Catch (TAC ceiling) allowed in accordance with respective AIs. The last 2 columns show the change in harvest relative to the 2009 PST catch agreement.

	SEAK Post season AI	SEAK TAC	NBC Post season AI	NBC TAC	WCVI post Season AI	WCVI TAC	Total BC TAC	BC TAC change relative to 2017	SEAK TAC change relative to 2017
2017	1.31	215,800	1.14	148,200	0.64	95,800	244,000		
2018	0.92	118,700	0.89	115,700	0.59	127,766	243,466		
2019	1.04	140,323[2] (137,200)	0.94	122,200 (122,200)	0.58	76,000 (86,840)	198,200 (209,040)	-5.18%	+2.27%
2020	1.11	140,323 (154,120)	1.16	141,700 (141,700)	0.67	78,500 (100,300)	220,200 (242,000)	-9.0%	-8.95%

Table 3. Allowable TAC vs observed catches in the AABM fisheries (SEAK, NBC and WCVI), percent reductions relative to 2017 catch ceilings under the 2009 PST agreement, and number of Chinook that weren't caught based on CTC catch and abundance models.

	SEAK TAC (TAC 2009 PST)	SEAK Obs. Catch	NBC TAC (TAC 2009 PST)	NBC Obs. Catch	WCVI TAC (TAC 2009 PST)	WCVI Obs. catch	BC obs. catch (TAC 2009 PST)	BC % catch change relative to 2009 TAC	Alaska % change relative to 2009 TAC	Total % change relative to 2009 TAC	# of Chinook remaining in water
2017	215,800	178,348	148,200	143,330	95,800	105,588	249,218 (211,500)	+17.83%	-17.35%	-0.48%	
2018	118,700	127,766	115,700	108,976	127,766	85,300	194,276 (243,466)	-20.2%	+7.63%	- 12.57%	39,994
2019	140,323[3] (137,200)	140, 307	122,200 (122,200)	88,026	76,000 (86,840)	73,482	161,508 (209040)	-22.7%	+2.27	-20.34%	44,425
2020	140,323 (154,120)	204,624	141,700 (141,700)	36,103	78,500 (100300)	45,581	81,684 (242,000)	-66%	+32.77%	-33.23%	109,802

[2] Catch ceiling not consistent with 2019 PST SEAK AI
[3] Catch ceiling not consistent with 2019 PST SEAK AI

List of documents submitted:

R.C. Lacy, R. Williams, E. Ashe, K.C. Balcomb III, L.J.N. Brent, C.W. Clark, D.P. Croft, D.A. Giles, M. MacDuffee & P.C. Paquet. 2017. Evaluating anthropogenic threats to endangered killer whales to inform effective recovery plans. *Scientific Reports.* 7: *14119* | DOI:10.1038/s41598-017-14471-0.

Murray, C.C., L.C. Hannah, T. Doniol-Valcroze, B.M. Wright, E.H. Stredulinsky, J. C. Nelson, A. Locke & R.C. Lacy. 2021. A cumulative effects model for population trajectories of resident killer whales in the Northeast Pacific. *Biological Conservation* 257. https://doi.org/10.1016/j.biocon.2021.109124.

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Evaluating anthropogenic threats to endangered killer whales to inform effective recovery plans

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Understanding cumulative effects of multiple threats is key to guiding effective management to conserve endangered species. The critically endangered, Southern Resident killer whale population of the northeastern Pacific Ocean provides a data-rich case to explore anthropogenic threats on population viability. Primary threats include: limitation of preferred prey, Chinook salmon; anthropogenic noise and disturbance, which reduce foraging efficiency; and high levels of stored contaminants, including PCBs. We constructed a population viability analysis to explore possible demographic trajectories and the relative importance of anthropogenic stressors. The population is fragile, with no growth projected under current conditions, and decline expected if new or increased threats are imposed. Improvements in fecundity and calf survival are needed to reach a conservation objective of 2.3% annual population growth. Prey limitation is the most important factor affecting population growth. However, to meet recovery targets through prey management alone, Chinook abundance would have to be sustained near the highest levels since the 1970s. The most optimistic mitigation of noise and contaminants would make the difference between a declining and increasing population, but would be insufficient to reach recovery targets. Reducing acoustic disturbance by 50% combined with increasing Chinook by 15% would allow the population to reach 2.3% growth.

Conservation science is tasked with quantifying the relative importance of multiple anthropogenic threats to species, both to determine if cumulative impacts exceed sustainable levels and to guide effective recovery plans[1–4]. However, cumulative human impacts are often poorly understood and inadequately addressed in conservation and management[5]. Fundamental research is still needed to integrate information on qualitatively different stressors into comprehensive models that reveal the cumulative impacts on measures of population growth, stability, and resilience[6]. Such work is needed, in part, because threats vary widely in their amenity to mitigation. When regulators require users to forego economic opportunities, it is important to have confidence that management actions will achieve the desired effect[7]. One way to accomplish this is to conduct "population viability analyses" (PVA) that use models of population dynamics to evaluate the relative importance of multiple anthropogenic stressors, singly and in combination, so that conservation can be directed toward efforts most likely to promote species recovery[8]. PVA can be a powerful tool for informing management and conservation decisions. However, the detailed population models used in PVA depend on: availability of estimates for demographic rates (both fecundity and survival and the variability in such rates); confidence that observed past rates are predictors of ongoing demography, or that trends can be foreseen; data for quantifying effects of threats on demographic rates; and a population model that adequately captures the key demographic, social, genetic, and environmental processes that drive the dynamics of the population of concern. Nevertheless, even when data on certain aspects of the population or its threats are not available, we can use PVA models to explore possible outcomes across a plausible range of values, and thereby identify which factors might be important and the target of additional research.

The Southern Resident killer whale (*Orcinus orca*, SRKW) population in the northeastern Pacific Ocean is one of the most critically endangered populations of marine mammals in the USA[9] and Canada[10]. The USA and

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Set	Scenario	Parameters varied	Population growth (r)
Baseline	Baseline	Rates as observed 1976–2015	−0.002
	Sensitivity Tests	See Supplementary Information (S.I.)	See S.I.
Individual Threats	Current	Chinook = 1.0; Noise = 85%; PCB = 2 ppm/y	−0.001
	Chinook	0.6 to 1.3 × baseline	−0.038 to +0.025
	Noise	0 to 100% of time	+0.017 to −0.004
	PCB	0 to 5 ppm/y	+0.003 to −0.008
Cumulative Threats	No Anthropogenic Threats	baseline Chinook; no noise, no PCB; no oil spills; no ship strikes	+0.019
	Low Development	25% decline in Chinook; 92.5% noise; low frequency oil spills and ship strikes (see Table 2)	−0.008
	High Development	50% decline in Chinook; 100% noise; higher frequency oil spills and ship strikes (see Table 2)	−0.017
Demographic Management	Fecundity	1 to 1.5 × baseline	+ 0.016
	Adult Mortality	1 to 0.5 × baseline	+ 0.009
	Calf Mortality	1 to 0.5 × baseline	+ 0.004
Threat Management	Chinook	1 to 1.3 × baseline	+ 0.025
	Noise	85% to 0%	+ 0.017
	PCB	2 to 0 ppm/y	+ 0.004
	Chinook & Noise	1 to 1.3 × Chinook; 42.5% Noise	+ 0.036

Table 1. Models of viability of the SRKW population for assessing current viability, sensitivity to anthropogenic threats, and responses to management. Population growth rates are mean r for Baseline, ranges for tests of Individual Threats, means for Cumulative Threat scenarios, and maxima for ranges tested in Demographic Management and Threat Management scenarios.

Canada have listed this transboundary population as Endangered, citing three primary risk factors: lack of the whales' preferred prey, Chinook salmon (*Oncorhynchus tshawytscha*); chronic and acute underwater noise and physical disturbance (e.g., from ferries, commercial ships, whale-watching boats, fishing boats, and recreational traffic); and high levels of contaminants, including polychlorinated biphenyls (PCBs)[10,11]. A recent Status Review[12] highlighted also the potential risk to this small, localized population from catastrophic events such as an oil spill. Governments and non-governmental organizations are currently seeking effective conservation measures for this high-profile population. Fortunately, the biological and environmental data available for SRKWs are rich by the standards of any marine mammal population. Long-term annual censuses, with continuous monitoring since 1976, coupled with the specialized diet, have allowed inference of quantitative relationships between prey and various metrics of fecundity and survival[13,14]. Thus, the prerequisites for a robust PVA suitable for guiding conservation are met.

PVA uses demographic models to assess risk to wildlife populations and evaluate the likely efficacy of protection measures, recovery targets, and restoration options[15,16]. We used the Vortex PVA model to examine the dynamics of SRKWs. Vortex[17–19] is a flexible, individual-based simulation that is freely available. Vortex has been used to set recovery goals and guide actions for many threatened species, including the Mexican wolf (*Canis lupus baileyi*)[20], Florida panther (*Puma concolor coryi*)[21], and Florida manatee (*Trichechus manatus latirostris*)[22]. Several recent PVAs on the SRKWs have shown how variability in demography[23] or inter annual variability in Chinook salmon abundance[12,24,25] could affect the population. We extend those approaches to consider also the sub-lethal effects of contaminants and acoustic disturbance, and the cumulative impacts of threats and interactions among them.

We first parameterized a Baseline model with demographic rates observed over 1976 through 2014, and tested the sensitivity of population growth to each demographic parameter. We then constructed one model that quantifies the population consequences of all three anthropogenic threats to SRKWs identified in Canadian[10] and USA[11] recovery plans. We compared the relative importance of each threat by projecting the population growth across the possible range of each threat. Finally, we used the PVA to explore the degree to which threats would have to be mitigated, alone or in combination, to reach a quantitative USA recovery target of sustained 2.3% growth over 28 years[11].

Results

Five sets of population models and the scenarios examined in each are listed in Table 1. The Baseline model projects mean population growth over the next 100 years of $r = -0.002$, with variation across years of $SD = 0.045$ (Fig. 1). These projections match very closely to the rate of $r = 0.002$, with $SD = 0.042$, observed over 1976 to 2014. The marginally lower growth in the model can be accounted for by future accumulation of low levels of inbreeding. After 100 years, the projected mean inbreeding coefficient is 0.067, about the same as results from mating between first-cousins. When inbreeding depression was eliminated from the Baseline model, the projected growth was $r = 0.002$, with $SD = 0.043$ – nearly identical growth and variation in growth to the trend in recent decades, and thereby confirming that the model replicates accurately the recent dynamics of the population.

Sensitivity tests of the influence of each demographic rate in the baseline PVA (Supplementary Information) show that, across the ranges of values tested, variation in fecundity (defined for the model as the mean proportion of adult females giving birth per year) accounts for most (77%) of the uncertainty in population growth rate.

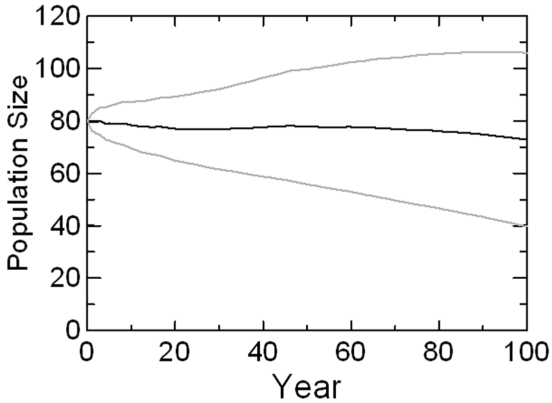

Figure 1. The distribution of 10,000 simulated trajectories with means and SD of the population size for northeastern Pacific Ocean SRKWs projected for 100 years, based on demographic rates observed from 1976 through 2014, applied to a starting population as it existed in 2015.

	Threats modelled					Population projection		
Scenario	Chinook trend	Noise	PCB (ppm/y)	Oil spill (big; small)	Ship strikes	Population growth (r)	Probability extinct	Probability final N < 30
No anthropogenic threats	constant	0	0	0	0	0.019	0	0
Current threats	constant	85%	2	0	0	−0.001	0	5%
Low increase	−25% in 100 y	92.5%	2	0.21%; 1.08%	1 per 10 y	−0.008	5%	31%
Higher increase	−50% in 100 y	100%	2	0.42%; 2.16%	2 per 10 y	−0.017	25%	70%

Table 2. Measures of viability of the SRKW population over 100 years under scenarios of minimal anthropogenic threats, current threats, and two levels of increased threats due to development. See text for explanation of threats modelled.

Annual adult mortality has some influence on the population trajectories (6%), but because mortality is already close to 0, there is comparatively less opportunity to improve the value of this parameter. Calf (first year) and juvenile (1 y to 10 y) mortality each accounted for about 3% of variation in population growth. Individual variation in reproductive success and temporal fluctuations (EV) in demographic rates had almost no effect on long-term population growth, as would be expected for a very long-lived species in which short-term fluctuations average out over time. Therefore, although our estimates of annual variation in rates are uncertain, refining the estimates would not change any conclusions about the effects of threats on the viability of the population. Given the small population size, inbreeding depression might cause sufficient adverse impact on population viability (6% of the total variance explained) such that it should not be ignored in assessments of long-term population viability. The impact of inbreeding was exacerbated slightly when we did not include avoidance of very close inbreeding (Supplemental Information).

Individual Threats. The set of models that includes estimates for the threats identified in the recovery plans – Chinook prey availability, noise and disturbance, and contaminants – was calibrated so that in the Current Threats scenario the demographic rates at existing threat levels reflect the mean demographic rates observed from 1976 through 2014. Thus, the Current Threats scenario mirrored the simpler Baseline scenario, except that rounding error in estimating effects of threats led to very slight deviation from the Baseline. The levels of these threats were then varied across broad ranges of values to determine which threat would have the greatest impact on population growth. Over the ranges tested, the effects of Chinook prey abundance on fecundity and survival had a greater effect on the population growth rate than did the other two factors (Fig. 2). Noise disturbance acts through decreased feeding efficiency in our model, but has a lesser effect than prey abundance because the maximum impact of boat noise 100% of the time would be to reduce foraging by about 20%. PCB accumulation rates that we tested result in mean levels in adult females of 0 to 132 ppm. Across this range, calf mortality is predicted to rise from about 7% to 50% (see Methods), and this impact shifts population growth from slightly positive to negative.

Cumulative Threats. Threats may interact, such that cumulative effects differ from those projected based on the summation of individual impacts. Full exploration of all of the possible interactions among the threats to the SRKW is not warranted at this time because individual threats are not yet well quantified. As more data on the above threats and other threats are acquired, management authorities can use the PVA framework to examine specific interactions of interest or full statistical analysis of all possible interactions[26]. To illustrate how cumulative threats can be assessed within the PVA model, we examined combinations of threat levels that represent the cumulative impacts of multiple threats for a few sample scenarios. We compared the Current Threats to a

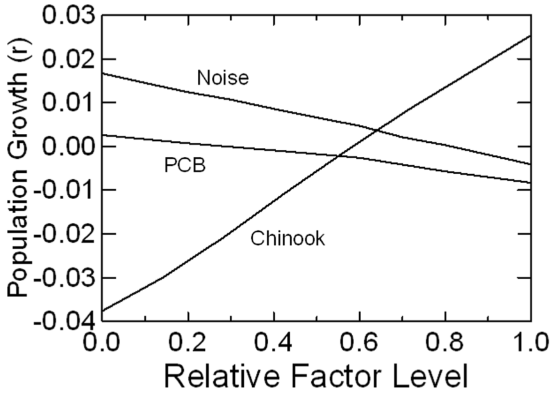

Figure 2. Effect of Chinook prey abundance (index varied from 0.60 to 1.30), noise and disturbance (boats present from 0% to 100% of time), and PCB contaminants (accumulation rate from 0 to 5 ppm/y) on mean population growth, while holding the other two factors at their baseline levels (1.0 prey index, 85% noise, and 2 ppm/y PCB accumulation). The x-axis is standardized to the range tested for each variable.

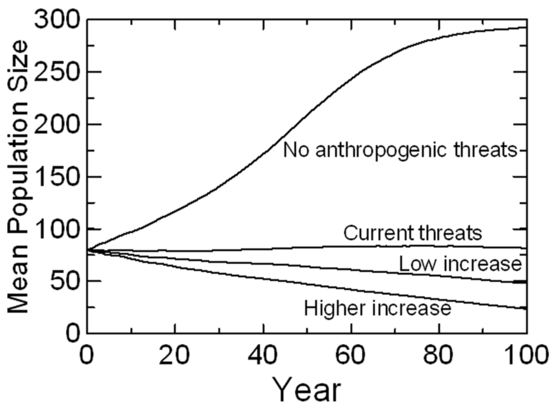

Figure 3. Mean projected SRKW population sizes for scenarios with (from top to bottom): no anthropogenic noise or contaminants; current Chinook abundance, noise, and PCBs; reduced Chinook, increased noise, and additional threats of oil spills and ship strikes as estimated for low level impacts of future industrial development; and these increased and additional threats with higher level impacts of development.

scenario with no anthropogenic threats and to scenarios with an increase in current threats and the addition of new threats. Figure 3 compares the population trajectory for the Current Threats with a scenario in which noise and PCB contamination were set to 0, and with two scenarios that describe levels of threat that could occur with proposed further industrial development and climate change. Table 2 shows the mean growth rates, probabilities of decline below 30 animals, and probabilities of extinction within 100 years under these scenarios.

The population could show robust growth if all anthropogenic threats were removed, but has no growth under current threat levels (Fig. 3). The combination of increased and additional threats expected under planned further industrial development in the habitat of the SRKW would cause population decline.

Demographic Management. The potential benefits of improvements in the primary demographic rates were examined in a set of Demographic Management scenarios. The demographic analyses indicate that reaching the SRKW recovery target of 2.3% growth is impossible by improving any single rate by a plausible amount, although increased fecundity would have the greatest positive influence on population growth (Fig. 4). To reach the recovery target, sustained mitigation of threats will be necessary to promote both increased fecundity and reduced mortality.

Threat Management. Improvements in demographic rates would need to be achieved by management actions that reduce threats or otherwise enhance the environment for SRKW. We therefore examined how population growth would respond to reductions in the levels of current threats. To achieve the recovery goal by increasing Chinook abundance alone would require a return to nearly the highest rates of Chinook abundance observed since 1979 (Fig. 5). If eliminating acoustic disturbance while maintaining current levels of Chinook abundance were possible, annual population growth could reach 1.7%. Removal of PCBs from the habitat would result in marginally positive (0.3%) growth, but the effect is much smaller than the impact of reduced noise and disturbance or increased Chinook abundance. Complete removal of both acoustic disturbance and PCBs is predicted to result in 1.9% growth. Therefore, reaching the recovery target without increasing Chinook salmon

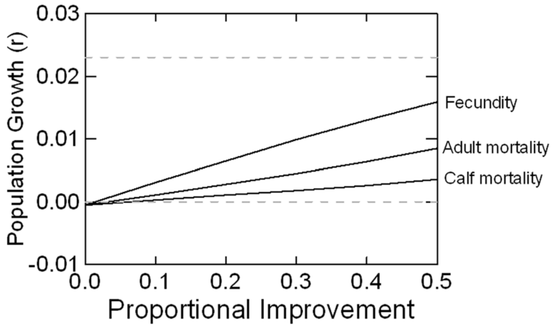

Figure 4. Mean population growth for SRKW achieved by improvements in demographic rates. Fecundity was increased from baseline to 1.5x baseline; mortality rates were decreased from baseline to 0.5x baseline. Dashed lines indicate a stated recovery target (2.3% growth) and r = 0.

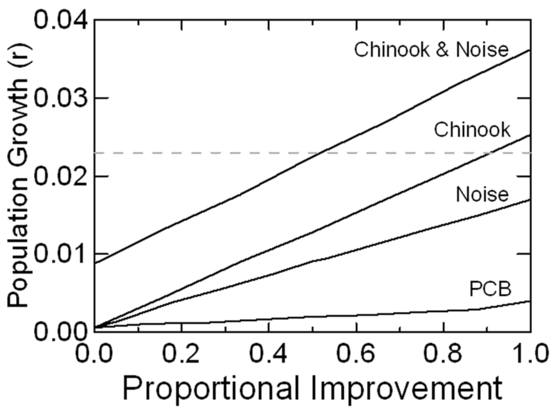

Figure 5. Mean population growth for SRKW achieved by mitigation of anthropogenic threats. Threat reductions are scaled on the x-axis from no reduction to the maximum reductions tested: Chinook abundance increased up to 1.3x the long-term mean; noise disturbance during feeding was reduced from 85% to 0; and PCBs were reduced from accumulation rates of 2 ppm/y to 0. The top line shows growth rates under a combination of varying levels of improved Chinook abundance plus mitigation of noise to half the current level.

numbers is likely impossible. Reducing acoustic disturbance by 50% and simultaneously increasing Chinook by more than 1.15x would allow the population to reach the 2.3% growth target. Other combinations of mitigation should be explored by management authorities as conservation options are identified.

Discussion

The SRKW population has experienced almost no population growth during the past four decades, and it declined in the last two decades. Intensive monitoring of the population since 1976 provides the information for construction of a detailed PVA model that closely replicates the observed population dynamics, and thereby provides a basis for projections under scenarios of increased anthropogenic threats or, conversely, increased mitigation actions. Models projecting population changes based on average demographic rates and fluctuations in those rates project that under the *status quo* the population will most likely remain near its current size. However, our use of baseline demographic rates averaged across 38 years of monitoring might give an overly optimistic projection for the SRKW if rates have deteriorated in recent years. A population projection based on demographic rates observed through 2011 projected a 1% annual mean growth[25], but a recent Status Review[12] projects a decline of 0.65% per year if demographic rates (such as recently lower fecundity) remain as they have been during 2011–2016. If ongoing monitoring indicates that these are not just short-term fluctuations in rates, then assessments of current viability, vulnerability to new or increased threats, and measures needed to achieve recovery will need to be revised.

When examined over ranges that encompass plausible improvements, the demographic parameter that presents the better opportunity for a large benefit to population growth is fecundity, rather than mortality. This finding is similar to a study of two bottlenose dolphin (*Tursiops aduncus*) populations off Australia, which found that variability in reproduction was more important than variability in mortality in driving differences between the populations[27]. There is simply more potential for improving reproduction than for improving adult survival when survival is already close to 1. Even complete elimination of adult mortality in the SRKW (not a biological possibility) would result in a population growth rate of 1.8%, still below the recovery goal of 2.3% growth. Although recovery cannot be achieved solely by improving adult survival, any decline in adult survival caused by new or exacerbated threats could have serious consequences for the population.

The PVA was useful for exploring scenarios representing the three main anthropogenic threats – prey limitation, acoustic and physical disturbance, and PCBs – that might worsen with increased development, or could be mitigated through management. Across the ranges of threat levels that we examined, reduction of the prey base was the single factor projected to have the largest effect on depressing population size and possibly leading to extinction, although either higher levels of noise and disturbance or higher levels of PCB contamination are sufficient to push the population from slow positive growth into decline. If additional threats from proposed and approved shipping developments (such as catastrophic and chronic oil spills, ship strikes, and increased vessel noise) combine with the predicted decline of Chinook due to climate change[28], then the population could decline by as much as 1.7% annually, have a 70% probability of declining to fewer than 30 animals, and have a 25% chance of complete extirpation within 100 years.

Mitigating multiple anthropogenic threats sufficiently to reach the recovery target will be difficult. The PVA is a useful way for managers to identify priorities for future research, and to focus conversations with ocean users and other special interests about the most pragmatic ways to promote recovery of endangered species. Those discussions must be integrated with considerations of feasibility, cost, societal impact, and timeframe for effective implementation. If a threat cannot be mitigated in a timescale relevant to conservation, or if costs are so high that they are prohibitive, thinking of those intractable problems as "fixed costs" in a cumulative impact management framework[4] might be useful. For example, our model results show that eliminating PCBs would provide less benefit to SRKWs than improving salmon returns or reducing anthropogenic noise and disturbance. This is fortuitous because imagining a way to eliminate PCBs that are persistent in the ecosystem is problematic[29], even though levels in tissues of SRKWs have been slowly declining in recent decades[30]. Identifying fixed costs that are difficult or impossible to mitigate allows a practical discussion about how to rank recovery actions among the anthropogenic factors that can be managed.

Of the three threats we considered, across wide but plausible ranges of each, salmon abundance is the greatest factor affecting SRKW population dynamics. Previously reported correlations of demographic rates with Chinook abundance[13,14,24] were used to parameterize our model, and Wasser *et al.*[31] recently offered insights into a mechanism that could cause the effect on fecundity: hormone levels indicate that SRKWs experience nutritional stress related to periods of lower abundance of Chinook prey and that this stress results in fewer successful pregnancies. Our PVA model estimated that SRKW recovery cannot be achieved without reaching the highest levels of salmon abundance observed since 1979, which was 30% higher Chinook salmon abundance than the long-term average between 1979 and 2008. This model result allows managers to focus discussions on whether achieving such a high sustained level of salmon abundance is attainable, and if so, how to achieve it. For example, removal of a hydroelectric dam on the Elwha River in the state of Washington is expected to increase spawning habitat for all five wild Pacific salmon species in the Salish Sea, but discussions about dam removal began in the 1960s[32] and the cost was in the hundreds of millions of US dollars. Restoration of spawning and rearing habitat could improve growth and survival of wild, juvenile salmon, but this takes political will, time, and money[33]. Improvement of marine survival of juvenile salmon might be possible by better management of net-pen salmon aquaculture sites that host and amplify viruses and parasites that have the potential to reduce survival of wild salmon[34,35]. Reducing Chinook harvest could provide an interim and strategic opportunity to rebuild depressed wild Chinook salmon runs and increase the number of Chinook available to whales in terminal areas like the Salish Sea[36]. Harvest reductions without longer term rebuilding plans might be an incomplete measure in places where Chinook harvests are already low due to abundance concerns or other constraints[37].

The SRKW population could be adversely affected by any new threats and further intensified impacts of the anthropogenic threats that we did assess. For example, pollutants other than PCBs might affect the population, and PCBs are known to have adverse effects beyond just reduced infant survival – such as reduced immune function[38]. However, other than calf survival, sufficient data are not yet available on the impacts of PCBs on demographic rates to allow incorporation of those threats in the population model. Moreover, threats to the population likely interact, perhaps in non-linear ways. For example, cetaceans that are food-limited might mobilize more lipids, and this will change the accumulated loads and harmful effects of PCBs and other organic pollutants. Similarly, reduction in foraging success because of boat noise might be of little consequence if prey is abundant, but could be critical if killer whales have difficulty procuring enough prey. If we can obtain data on additional threats and the interactions among threats, such effects could be included in the PVA models. At present, given that only estimates of approximate average effects of some threats are included in the model, inclusion of higher level interactions is premature.

While acknowledging that we examined only the identified primary threats to the SRKWs and that we cannot yet fully assess possible complex interactions among those threats, an important finding from our PVA is that reaching the recovery target will likely require mitigation of multiple threats. For example, the PVA projects that a 50% noise reduction plus a 15% increase in Chinook would allow the population to reach the 2.3% growth target. Noise is a particularly attractive issue to address in a management context, because it is amenable to several possible mitigation scenarios[39,40]. With respect to noise from commercial shipping, preliminary calculations suggest that the distribution of source levels of individual ships follows a power law, implying that quieting the noisiest ships will reduce overall noise levels by a disproportionate amount[41]. Identifying the noisiest ships operating in SRKW critical habitat[42] and creating incentives to reduce their noise outputs through speed restrictions and maintenance might generate considerable reductions in noise levels. The International Maritime Organization and the International Whaling Commission have urged nations to reduce the contribution of shipping to ocean ambient noise, with some countries adopting a pledge to reduce anthropogenic noise levels by 50% in the next decade[43]. However, from the perspective of a foraging killer whale that emits high-frequency (18-32 kHz) echolocation clicks to detect and capture salmon, high-frequency noise from small, outboard vessels that follow whales might cause a greater reduction in a killer whale's foraging success than low-frequency (<1 kHz) background noise from commercial shipping[44].

Clearly, even without new or increased external threats, the SRKW population has no scope to withstand additional pressures. The current situation for SRKWs gives little cause for optimism. This is likely to worsen, given the energy-related project proposals already approved for the region[45], which will increase broadband ocean noise levels and the risk of ship strikes and oil spills[46]. Our models of the additional threats expected with a proposed increase in oil shipping show that these threats will push a fragile population into steady decline. Obviously, countering such additional threats sufficiently to achieve SRKW population recovery would require even more aggressive mitigation actions than if there were no such increasing threats to the population.

The case study we present offers an unusual opportunity to examine multiple anthropogenic threats in a wildlife population that is extremely data-rich by the standard of any marine ecology study[5]. One threat (the impact of prey abundance through the prey-demography link) has been well studied for decades. Another (acoustic disturbance) is relatively well appreciated in that there are documented relationships between higher noise level and reduction in foraging success. However, a conceptual step is required to convert the reduction in foraging to a reduction in prey acquisition. Full consideration of noise impacts would need to include complex interactions among reduced foraging time, reduced detection space, and reductions in prey availability. The third kind of threat (population consequences of PCBs and other persistent pollutants) relies on very few data points to calibrate the effect of the PCBs only on whale calf survival, which underestimates the total population consequences of contaminants in two ways. Lack of concentration-response studies on compounds other than PCBs hinder our ability to model population consequences of PBDEs or other contaminants. Similarly, existing studies do not allow us to predict effects of contaminants on pregnancy rate or adult mortality. This spectrum of data-rich to data-poor steps in predicting population consequences of multiple stressors is ubiquitous in conservation and ecological studies[2,47]. The funding to fill knowledge gaps with empirical data may be lacking, or in the case of critically endangered species, time to wait for science to fill data gaps may be insufficient[48]. Some authors use expert elicitation[49,50] to fill data gaps. Expert opinion or examination of hypothetical, but plausible scenarios should be used to augment rather than replace the available data.

The case study presented here illustrates the use of PVA as a method to inform difficult conservation decisions, by simulating across plausible ranges of uncertainty. For example, sensitivity analyses revealed that some factors (e.g., individual variability in breeding success) have no effect, and such knowledge gaps should not be a barrier to management action. Given our inability to manage some insidious threats, such as persistent organic pollutants that are already in the environment, it is reassuring that the model predicts that this stressor has the smallest adverse impact on the population, at least via the pathway of reduced calf survival. The PVA can focus priority research on questions that make a practical difference. Studies of foraging efficiency under varying levels of anthropogenic disturbance are needed only because the population is prey-limited. If doubling Chinook salmon numbers were possible, and returning them to levels seen in the 1920s[51], consideration of other anthropogenic impacts on the whales' foraging efficiency might not be necessary. Alas, this is not a realistic scenario, and the model therefore points to the importance of including both improvement in prey abundance and reduction in noise as the more effective mitigation pathway.

Unfortunately, focus on only the immediate, tractable threats is all too common in conservation. For example, conservation of grizzly bears (*Ursus arctos horribilis*) in the continental United States focuses on roads and development activities, but the primary concern is that the species has been absent from most of its range since the 1800s[52]. Similarly, the current small size of the SRKW population was not caused by lack of salmon. The whales' depleted status is due in large part to the legacy of an unsustainable live-capture fishery for display in aquariums[53]. Salmon, noise, and contaminants are important factors that can prevent recovery. Many policies, including the US National Environmental Policy Act, require regulators to consider the effect of a proposed activity "which results from the incremental impact of the action when added to other past, present, and reasonably foreseeable future actions regardless of what agency (Federal or non-Federal) or person undertakes such other actions (40 CFR § 1508.7)." Allocating impacts among multiple ocean user sectors may be difficult, but in the case study we present, the population is sufficiently imperilled that it has little or no scope for tolerating additional stressors.

Methods

The SRKW population is closed to immigration and emigration, every individual in the population is known, and the population has been censused annually for decades[11]. Individuals were identified by their unique fin shapes, saddle patches, and the presence of any nicks or scratches, and sexed using distinctive pigmentation patterns around the genital slits. Male and female offspring remain within the natal, matrilineal unit, although mating occurs within and between these pods. The term "resident" refers to their residency in inshore waters of southern British Columbia (Canada) and Washington state (USA) in the summer months, when they feed almost exclusively on Chinook salmon[13,14,54,55]. Given that there is no dispersal from the population[56], mortality was recorded if an individual's matriline was observed in the population within a year but the individual did not appear.

We used values of demographic parameters calculated from the census data to build the population model in the Vortex PVA program[17,18]. We included temporal variation in demographic rates ("environmental variation"), based on inter-annual variability in parameters observed since 1976, and we included individual variation in age of maturity and probability of reproductive success. The Vortex simulation model of possible future population trajectories includes demographic stochasticity (binomial variation in individual fates); random assignment of sex and a bi-sexual mating system, resulting in fluctuations in sex ratio and mate availability that can affect small populations; and projections of loss of genetic diversity, allowing for inclusion of inbreeding depression. We quantified population growth as the mean exponential rate of increase ($r = \ln[N_{t+1}/N_t]$).

Modelling was conducted in stages. First, a "Baseline" model was developed to represent the population trajectories if demographic rates remain the same as have been observed in recent decades. We confirmed this Baseline model by comparing simulated dynamics with recent population trends. Secondly, we conducted sensitivity tests on uncertain demographic rates in the model to determine which parameters had large effects on the projected population growth. Thirdly, we used a set of models of Individual Threats that tested ranges of values for the primary threats identified in the recovery plans to determine which would have the greatest effects on population projections. Fourthly, we examined Cumulative Threats scenarios to project the fate of the population if further industrial development increases existing threats and adds new ones. A set of Demographic Management scenarios was then examined to determine the population growth that could be achieved by improvements in demographic rates. Finally, we explored Threat Management scenarios to assess the plausibility of reaching sustained annual population growth of 2.3% given various options for increasing salmon abundance, reducing ocean noise levels, or reducing contaminant levels. The following section describes key parameter estimates used in the model. More detailed description of the modelling methods is presented in Supplementary Information. The input files for the Vortex project are available at http://www.vortex10.org/SRKW.zip and from the Dryad Digital Repository at https://doi.org/10.5061/dryad.46vq7.

Baseline PVA. We started the simulations with the ages, sexes, and pod membership of the killer whales living in 2015. We specified the mother of each animal, where known (for 76 of 80 living animals)[57]. Based on previous genetic data on paternity[58], we specified in the simulation that females would not mate with their father, a son, or a maternal half-sibling. What effect lower levels of inbreeding or the inevitable accumulated inbreeding in a closed population will have on any cetacean is unknown. We modelled inbreeding depression as being caused by recessive lethal alleles, with 6.29 "lethal equivalents" (the negative of the slope of log(recruitment) against the inbreeding coefficient), the mean combined effect of inbreeding on fecundity and first-year survival in a survey of impacts on wild species[59].

Demographic rates were calculated from individual animal histories compiled by the Center for Whale Research[57], using data collected from 1976 through 2014. The time series begins when the population was depleted by live-captures for display in aquariums[60]. The time series therefore includes periods of moderate population growth (1976 to 1993), subsequent decline, and approximate stability. Demographic rates were estimated for the age-class groupings used in recent models[24,61], except that we set an upper limit for female breeding at 45 y rather than 50 y, because no females in the population have been documented to produce calves at older ages. Thus, we calculated survival and (for adult females) fecundity rates for calves (first year), juveniles (defined as from 1 y through 9 y of age), young mature females (10–30 y), older reproductive females (31–45 y), post-reproductive females (46 y and older), young mature males (10–21 y), and older males (22 y and older). Killer whales can survive many years after reproductive senescence, but estimating maximum longevity is difficult in such a long-lived species[62]. We set an upper limit of age to 90 y in our models, although only about 2% of females would be expected to reach this age, and only about 2% of males (with higher mortality) would be expected to exceed 50 y. Females stop breeding long before the maximum age, so the long-term population growth would not be affected by the upper age limit unless post-reproductive females benefit the pod in ways other than through their own reproduction.

Mortality for each age-sex class was averaged across the 39 years of data to obtain mean annual rates. We did not try to partition observed mortality into presumed causes of death. The use of these historic data for our Baseline model makes the implicit assumption that the frequency of deaths due to the various causes remains the same as has been observed across recent decades. The variation in mortality observed across years has two components: 1) environmental variation (fluctuations in the probability of survival), and 2) demographic stochasticity (binomial variation in individual fates). To determine how much of the observed variation was due to environmental variation, the variance due to demographic stochasticity can be calculated from the expectation for a binomial process, and then subtracted from the total variation across years. Calculated annual mortality rates (and environmental variation) ranged from a low of 0.97% (SD = 0) for young adult females to 17.48% (SD = 17.96) for calves. Although the lack of evidence for annual variation in the mortality adult females beyond that expected from random sampling of a constant probability might seem optimistic, for long-lived species a low level of annual variation in rates would have negligible effect on long-term population trajectories. We confirmed through sensitivity tests (Supplementary Information) that the environmental variation entered into the population model has no effect on our results.

The breeding system is polygamous, with some males able to obtain multiple mates, females mating with different males over their lifetimes, and mating between and within pods. Males become sexually mature (actively breeding, which may occur several years after they are physiologically capable of breeding) from 12 to 18 y of age. Thus, in the model, each male was assigned an age of sexual maturity by randomly selecting a value from 12 to 18. Variance in reproductive success among individual females and males will cause genetic diversity to be depleted faster and inbreeding to accumulate faster than would occur if mating was assumed random. Information is available on male mating success[51], and we incorporated variation in male and female reproductive success in the model (Supplementary Information). Our models project an effective population size that is 37% of the total size, close to an estimate obtained from genetic data[58].

Breeding rates, expressed as the proportion of the females of an age class that produce a calf each year, were calculated from annual census data. Rates ranged from 0% for post-reproductive females (age >45 y), to 7.88% (SD = 4.15) for older adult females (age 31–45 y), to 12.04% (SD = 3.54) for young adult females (age 10–30 y).

The upper limit on population size was set to 300, so that carrying capacity (K) would not restrict future population growth except under the best conditions tested. In the projections of current or expected conditions, the SRKW populations never reached this limiting size, and rarely exceeded 150 animals in any of the independent iterations of each simulation. Population recovery was assessed by the mean growth rate each year calculated before any carrying capacity truncation. Thus, the growth rate reflects the demographic potential and is not affected by the limit on population size in the model.

The SRKW population was projected for 100 years. For the initial exploration of parameter uncertainty, the simulation was repeated in 10,000 independent iterations to obtain high precision in mean and variance estimations. For comparisons among alternative management scenarios, less iteration is needed to obtain the relative influence of input values, and tests were run with 1,000 iterations. Sensitivity tests were conducted by varying each basic demographic rate (life table values for fecundity and mortality) over a range of $\pm$ 10% around the baseline value. For several model variables that describe other aspects of the population dynamics and are also very uncertain, a wider range of values was tested (see Supplementary Information).

Individual Threats. We explored the effects of three threats identified in the recovery strategies. For each of prey abundance, noise disturbance, and PCB contaminants, we scaled impacts such that the estimated current level of the threat resulted in the mean demographic rates reported over recent decades. Effects of prey limitation were modelled using published relationships linking inter-annual variability in Chinook salmon to inter-annual variability in calf and adult mortality[63] and fecundity[13,61]. A prey index was calculated by dividing the total salmon abundance in each year by its average abundance over the 1979–2008 period[63]. The relationship of mortality to prey abundance was modelled with a multiplier of baseline mortality that is a linear function scaled to 1 when salmon abundance was at the mean observed level over period of observation: MortalityFactor = 3.0412 – 2.0412 * PreyIndex. The relationship of birth rate to prey was modelled with logistic functions, with the intercept scaled to yield the observed birth rates for young females (12.04%) and older females (7.88%) when PreyIndex = 1. For relationships of form BirthRate = exp(A + B*PreyIndex)/[1 + exp(A + B*PreyIndex)], the function parameters were A = −3.0 and B = 1.0 for young females, and A = −3.46 and B = 1.0 for older females. (See Supplementary Information for more details on these relationships.) To explore the impacts of prey abundance across a range of plausible values, we varied the prey index from approximately the lowest level (0.60) reported since 1978 to approximately the highest level (1.30).

Effects of noise on demography were modelled using the approach outlined in previous analyses of loss of acoustic communication space[4,64]. We used summertime observations to estimate the proportion of time boats were present (during daylight hours) while the whales were foraging and the reduction in foraging expected with that amount of acoustic disturbance. We calibrated the model of noise impacts so that the mean Baseline demographic rates are obtained at the reported level of disturbance. We then simulated the relative change in foraging time and consequently demographic rates across the spectrum from no noise impact at all, to the upper limit expected if boat disturbance increased from current, already high, levels to 100% of time. We do not have data on the amount of acoustic disturbance in the winter feeding areas, but the modelling based on observed summertime disturbance provides a means to project a range of population consequences if changes in disturbance overall mirror those that are possible in the summertime habitat. Land-based observations have shown that SRKWs reduce their time spent feeding in the presence of boats by 25%[65]. Vessels are present 85% of the daytime, and SRKWs are foraging in the presence of vessels an estimated 78% of that time. Thus, for the 85% current (baseline) exposure to vessels, feeding is expected to be reduced by 16.6% (= 85% × 78% × 25%) due to disturbance by boats. To translate the reduction in feeding into its demographic consequences, we multiplied the prey index by a factor of (1 − 0.195 * Noise)/(1 – 0.166) to obtain the proportional availability of prey. This proportion is thus 1 in the current, baseline conditions (Noise = 0.85), 0.965 when vessels are always present (Noise = 1.00), and 1.20 assuming no disturbance from vessels. The noise-modified index of prey availability was then used to determine the consequent mortality and fecundity rates. We recognize that anthropogenic noise can also have less direct effects on wildlife, including disruption of social behaviours and even impeding responsiveness to other sensory modalities[66].

Our model of accumulation, depuration, and impact on calf survival of PCBs was based on the approach described by Hall *et al.*[67,68] with modifications in rates for SRKW[69]. Calves obtain their initial load of contaminants from their dams through gestation and lactation, and females producing calves thereby depurate an estimated 77% of their contaminants[67]. Otherwise, males and non-breeding females accumulate PCBs in the blubber of at a rate that we varied from 0 to 5 ppm/y in our tests. Few data are available on PCBs in the SRKW population with which to calibrate the model of PCB bioaccumulation, and the levels of PCBs reported in SRKW might have been dropping slowly in recent years. Reported levels in adult female SRKW range from 55 ± 19 ppm sampled in 1993–1996, 37 ± 42 ppm sampled in 2004–2007, and 30 ± 31 ppm sampled in 2008–2013[30]. Our population model generates a mean 28, 55, and 81 ppm PCBs in adult females when bioaccumulation rate is 1, 2, and 3 ppm/y, respectively. Effects of maternal PCB load on calf mortality were modelled using a logistic response function (survival = exp(2.65 − 0.02 * PCB)/[1 + exp(2.65 − 0.02 * PCB)]), fitted to the two observed data points for SRKW (survival = 0.8252) and the nearby northern resident killer whales (survival = 0.9218)[24], with the mean PCB levels (55.4 ppm and 9.3 ppm, respectively)[70] reported from the time period in the middle of the span over which mortality rates were calculated. If we use the more recent, lower estimates of PCB loads in SRKW to estimate the impacts, our response function would have a steeper slope. There are not yet sufficient data on effects of PCBs on other demographic rates to allow inclusion of any other effects of PCBs (or other contaminants) in our PVA model.

Cumulative Threats. We modelled two scenarios to represent the cumulative impacts of possible increases in threats, based in part on a recent environmental impact assessment submitted to Canada National Energy Board[45] evaluating effects of a proposed oil pipeline and associated tanker traffic. For the purposes of this PVA, projected increases in anthropogenic threats are not meant to mimic any one industrial development, but rather a general process of industrialization reflecting the number of port expansions, pipeline proposals, and liquefied natural gas terminal proposals pending for the BC coast[4]. For a low level scenario, we used the catastrophe option in Vortex to add the possibility of large (>16,500 m^3) and smaller (>8,250 m^3) oil spills. The frequencies of a big spill (0.21% chance per year) and a smaller spill (1.08%) were based on an industry projection of the likelihood of such spills caused by proposed increase in tanker traffic[71]. Based on the percent overlap of oil coverage and critical habitat, we estimate that if a large oil spill were to occur, about 50% of the SRKWs would be killed due to direct exposure to the oil. We estimate that 12.5% of the SRKWs would be killed by exposure to oil from a smaller spill. For a scenario with higher level impacts of development, we doubled the frequency of oil spills.

These energy development scenarios also included an increase in vessel noise and disturbance of feeding, with the current vessel presence of 85% of time increased to 92.5% in the low level scenario and to 100% in the high level scenario. We also included a probability of additional deaths of killer whales due to ship strikes, with one death per decade in the low level and two deaths per decade in the high level scenario. Although some persistent organic pollutants might increase under increased industrial activity in the SRKW habitat, PCBs have been phased out of production and are in decline in at least some fish species in low-development basins[72]. Lacking data on likely long-term trends in the contaminant loads of SRKW prey, we did not include any change in such pollutants in these scenarios.

Climate change is projected to cause a decline in Chinook abundance[28], and we modelled this possibility with a projected 25% (low scenario) or 50% (high scenario) decrease in Chinook over the next 100 years.

Demographic Management and Threat Management scenarios. We used the PVA to simulate how much improvement in demographic parameters or how much reduction in anthropogenic threats, singly or in combination, would be required to reach a stated recovery objective of sustained annual population growth of 2.3% for 28 years[11]. In calculating the growth for these models, we started the tally 20 years into the simulation to avoid short-term demographic fluctuations as the age structure adjusts to new demographic rates, and growth was tallied over the subsequent 28 years. For the set of Demographic Management scenarios, we assessed the relationship between improved demography and population growth. Birth rate was incremented by 1.1x, 1.2x, 1.3x, 1.4x, and 1.5x, whereas calf mortality and adult mortality were decreased by 0.9x, 0.8x, 0.7x, 0.6x, and 0.5x. Next, in Threat Management scenarios, we modelled the effects of reduced threats, with the consequences resulting from the functional relationships to demography. We increased salmon abundance (up to the highest level of the Chinook index observed between 1979 and 2008, namely 1.3 times the long-term average). We simulated the improved demography if acoustic disturbance were reduced or eliminated. We considered the population consequences of improved calf survival resulting from reduction of PCBs, testing rates of future accumulation in SRKW from the estimated current 2 ppm/y to down to 0 ppm/y. Finally, we tested scenarios that both reduced acoustic disturbance by half and increased salmon abundance up to 1.3x.

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Author Contributions
P.P. coordinated the project. K.C.B. and D.A.G. provide the core census and demographic database. R.W., E.A., L.J.N.B., C.W.C., D.P.C., P.P., and M.M. provided data on threats. R.C.L. built the population model and conducted the simulations. R.W., R.C.L. and P.P. led the writing. All authors contributed to and reviewed the manuscript.

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A cumulative effects model for population trajectories of resident killer whales in the Northeast Pacific

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ABSTRACT

Like numerous species at risk, the resident killer whale populations of the Northeast Pacific are vulnerable to the cumulative effects of anthropogenic threats. A Pathways of Effects conceptual model summarised the current understanding of each threat (prey availability, acoustic and physical disturbance, and contaminants), threat interactions, and potential impacts to fecundity and mortality. A Population Viability Analysis utilised the most recent available data to quantify impacts of threats on population parameters. The impacts of individual and cumulative threat scenarios on modelled Southern and Northern Resident Killer Whale populations were compared to the observed population demographics to define a model that best captured the real world dynamics. Of the individual and combined threat models tested, the cumulative model incorporating all threats predicted demographic rates closest to those observed for both populations. Recent low Chinook salmon abundance and its interactions with vessel disturbance and contamination strongly influenced modelled killer whale population dynamics. The cumulative effects population viability analysis model projected a mean increase in the modelled Northern Resident Killer Whale population to the carrying capacity within 25 years. In contrast, the mean modelled Southern Resident Killer Whale population trajectory was projected to decline under current conditions, with a 26% probability of population extinction, and in those projections, extinction was estimated to occur after 86 (± 11) years. Our results highlight the importance of considering the collective impact of multiple threats to imperilled species and the necessity of testing management and mitigation measures aimed at recovery using a holistic, validated model.

1. Introduction

Effective conservation of species at risk now requires an understanding of the cumulative effects of multiple activities in the ecosystem. The impact of a single threat on a species through time and across its geographical range will always have a degree of associated uncertainty and these uncertainties are compounded when multiple threats co-occur and potentially interact. Uncertainties and interactions among threats make recovery efforts fraught with confusion as decision-makers must consider all evidence to assess potential recovery actions. This balance is demonstrated by killer whale populations around the world that are under threat from several anthropogenic pressures (Desforges et al., 2018; NASEM, 2017). The Southern Resident Killer Whale (SRKW) population is listed as Endangered under the Canadian Species at Risk Act (SARA) and the US Endangered Species Act, and the more northerly but sympatric Northern Resident Killer Whale (NRKW) population is listed as Threatened under the SARA. All populations of Resident Killer Whales are piscivorous, feeding primarily on Chinook (*Oncorhynchus tshawytscha*) and chum salmon (*O. keta*), but despite the similarities in diet and a substantial overlap in range (from southeastern Alaska to Washington State), SRKW and NRKW do not interact with one another socially and are distinct in terms of their culture, acoustics, and genetics (Ford et al., 1998, 2000; DFO, 2017a, 2017b). A comparison of their population dynamics can thus provide insights into the different ways they are affected by human threats (NASEM, 2017).

Long-term photo-identification census surveys for SRKW and NRKW, which were initiated in the 1970s and continue to the present day, show contrasting trends in the two populations (DFO Cetacean Research

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Program; Center for Whale Research, CWR). Population trends based on these census data indicate that the SRKW population has experienced an overall negative population growth rate (−0.002; 1979–2017), with a particularly sharp decline between 1995 and 2001 (Fig. 1). Since then, the SRKW population has shown little recovery, having 73 members as of July 2019. In contrast, the NRKW population has experienced a steady increase over the census period (population growth rate = 0.02; 1979–2017), except for a decline between 1997 and 2001 (Fig. 1). The NRKW population has since increased from 219 members in 2004, to 310 members in 2019 (Olesiuk et al., 2005; DFO, 2020).

The three primary threats to SRKW and NRKW are reduced prey availability, acoustic and physical disturbance, and environmental contaminants (COSEWIC, 2008; Ford et al., 2010; DFO, 2011, 2017a; NMFS, 2008). There is strong evidence that survival and fecundity of these populations are affected by prey availability (Ford et al., 2010; Ward et al., 2009) but limited quantitative evidence on the impacts of disturbance and contaminants. These threats may act on the populations at multiple life history stages and throughout their range. Thus, there is potential for cumulative effects on these populations through repeated exposures to a single threat, exposures at multiple life stages to a threat, and/or exposures to multiple threats. Additionally, threat interactions are known to be common when multiple stressors act within a system (Crain et al., 2008; Darling and Côté, 2008) and non-linear relationships make the effects at a population level difficult to determine. Understanding the effects of cumulative as well as individual threats is therefore necessary to inform the development of effective population conservation strategies (NASEM, 2017).

Several approaches have been used for consideration of cumulative effects on cetaceans. Previous cumulative effects assessments (CEAs) fall into three categories: risk assessment, statistical analysis, and population viability analysis (PVA) (Lacy et al., 2017; Lawson and Lesage, 2012; O et al., 2015). Risk assessment has been used to rank threats and activities of interest occurring in cetacean habitat (Lawson and Lesage, 2012; O et al., 2015). Statistical models have been used to evaluate the impact of single threats on mortality and fecundity of resident killer whales (Vélez-Espino et al., 2015; Ward et al., 2009). A PVA model was developed to evaluate cumulative effects of anthropogenic threats on the SRKW population (Lacy et al., 2017).

The aim of the current assessment is to define and apply a cumulative effects model to evaluate and compare the individual and cumulative effects of anthropogenic threats on both the SRKW and NRKW populations of resident killer whales (after Lacy et al., 2017; Clarke Murray et al., 2019). The study is limited to considering the primary threats identified in the SARA action plan for NRKW and SRKW (COSEWIC, 2008; Ford et al., 2010; DFO, 2011, 2017a, 2018; NMFS, 2008). The definition and testing of an acceptable cumulative effects model will support evaluation of future changes in anthropogenic activities and potential mitigation measures and management actions.

2. Methods

2.1. Cumulative effects assessment

The cumulative effects assessment consisted of two phases: a Pathways of Effects (PoE) conceptual model and a Population Viability Analysis (PVA) quantitative simulation model. The PoE conceptual model described the impacts of threats (or stressors) on killer whale vital rates (mortality and fecundity). As the interaction of threats over space and time can alter their respective intensities and consequent effects on individuals and populations, potential interactions between threats were also assessed to more accurately represent the natural system. The PoE conceptual model consisted of a visual representation of the threat linkage pathways, with supporting justification text (Stephenson and Hartwig, 2009; Government of Canada, 2012) and was developed through literature review and elicitation of expert opinion through consultation with colleagues in relevant fields of expertise.

The outputs of the PoE conceptual model were used to design and refine the structure and parameterisation of the PVA model, building upon the methods and results of previous work (DFO, 2018; Lacy et al., 2017; Taylor and Plater, 2000; Vélez-Espino et al., 2015; Ward et al., 2009; Williams et al., 2017). Existing literature and data were used to parameterise the impact of each threat on killer whale vital rates, and previously published relationships were updated with recent data and re-analysed (detailed in Supplementary Material). The quantitative values and relationships specific to each population (SRKW and NRKW) were used to define the inputs to the population model describing the combined impact on population persistence through time. The model structure builds upon an existing PVA model developed for the SRKW population by Lacy et al. (2017). To capture the unique population structure and threat exposure, a PVA model was developed for each population (SRKW and NRKW) using Vortex 10.3.1, an open access modelling software (Lacy and Pollak, 2014).

2.2. Killer whale population model

Population models were constructed using census data obtained from DFO's Cetacean Research Program encompassing the years 1979–2017 (DFO, 2020; DFO CRP, unpublished data). Annual population surveys have occurred without interruption since 1973 for the NRKW population and 1976 for the SRKW population (DFO Cetacean Research Program; Center for Whale Research, CWR). The SRKW census is considered to be more precise than the NRKW census, as not all members of the Northern

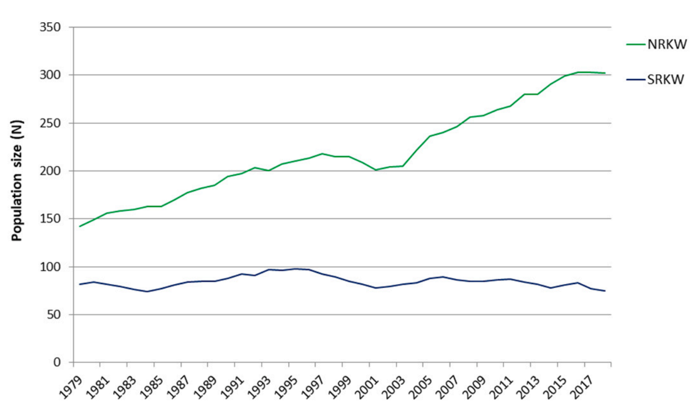

Fig. 1. Resident killer whale population time series, 1979–2018. Source data: long-term photo-identification census surveys for Southern and Northern Resident Killer Whales (SRKW and NRKW), which were initiated in the 1970s and continue to the present day (Fisheries and Oceans Canada Cetacean Research Program; Center for Whale Research).

population have been seen each year (DFO, 2018, 2020). By using demographic rates starting in 1979 for both populations, the time series is composed mostly of data from direct observations rather than reconstructed data (Olesiuk et al., 2005). The killer whale reproductive system was defined as polygynous and sexually dimorphic with observed population parameters (Table A1; Olesiuk et al., 2005; Vélez-Espino et al., 2014; Ward et al., 2009). Details can be found in Appendix A. The SRKW range extends from southeastern Alaska to central California and the NRKW range from the coastal waters of Glacier Bay (Alaska, USA) to Gray's Harbor (Washington State, USA) (Ford, 2006; Ford et al., 2000).

The neighboring Southern Alaska Resident Killer Whale (SARKW) population has a similar life history strategy but is relatively removed from the threats to which the SRKW and NRKW populations are exposed. The SARKW population has over 700 individuals and has an annual rate of increase considered to be at its maximum, 3.5%, attributed to rebounding salmon stocks (Matkin et al., 2014). The SARKW census data, which began in 1984, were used to define the reference mortality and fecundity rates for the population models, assuming the rates represent those expected from a population in unrestrained growth. The SARKW population is not considered to be pristine as it is exposed to anthropogenic impacts; contaminants and oil spills are the main threats (Matkin et al., 1998, 2014). Notably, a major oil spill (*Exxon Valdez*) occurred in 1989 and resulted in a 33% loss of the resident AB matriline (Matkin et al., 1998, 2008), which has not recovered. However, these anomalous deaths were excluded from the data analysis of Matkin et al. (2008) and do not affect the estimates of vital rates for SARKW used in the present analysis (Table 1). The rates and age/sex structure of the SARKW population were found to be similar to NRKW in their period of unrestrained growth, except that the age of maturity was one year younger for SARKW (Matkin et al., 2008; Olesiuk et al., 2005).

The SARKW vital rates (Table 1) were used in the SRKW and NRKW population models to represent the reference vital rates that determine the growth of each population in the absence of anthropogenic threats. This is an important change from the SRKW model developed by Lacy et al. (2017), where the "baseline" was defined using the mean demographic rates that were observed from recent decades and would therefore include current threat levels. Model scenarios were developed and tested for individual and cumulative threats where threats (described in further detail in later sections) were included in the model as modifiers of the SARKW reference vital rates.

Table 1
Age-specific mortality and fecundity rate for each Resident Killer Whale population: Southern Resident Killer Whales (SRKW), Northern Resident Killer Whales (NRKW) and Southern Alaska Resident Killer Whales (SARKW). SRKW and NRKW data were sourced from Vélez-Espino et al. (2014) for years 1987–2011. SARKW data were sourced from Matkin et al. (2014) for years 1984–2010.

	SRKW	NRKW	SARKW
Age class (y) (male and female combined)	Age-specific mortality rate		
0–1	0.215 (SD = 0.284)	0.078 (SD = 0.082)	0.054 (SD = 0.244)
1–2	0.019 (SD = 0.047)	0.028 (SD = 0.019)	0.003 (SD = 0.040)
2–5	0.019 (SD = 0.047)	0.028 (SD = 0.019)	0.010 (SD = 0.054)
6–10	0.019 (SD = 0.047)	0.028 (SD = 0.019)	0.012 (SD = 0.064)
10–16	0.015 (SD = 0.033)	0.011 (SD = 0.012)	0.008 (SD = 0.032)
17–51	0.033 (SD = 0.054)	0.011 (SD = 0.025)	0.023 (SD = 0.066)
51+	0.072 (SD = 0.108)	0.117 (SD = 0.114)	0.217 (SD = 0.292)
Age class (y) (female only)	Age-specific fecundity rate		
10–30	0.116 (SD = 0.077)	0.142 (SD = 0.046)	0.233 (SD = 0.118)
31–50	0.069 (SD = 0.074)	0.101 (SD = 0.051)	0.154 (SD = 0.118)

2.3. Population viability analysis modelling

The population genealogical and demographic data were partitioned to allow model validation and verification; the complete set of living animals in the year 2000, with their known dams, calving histories, and genealogies, were used as the starting population for each of the population models (SRKW and NRKW). This allowed a comparison of the modelled and observed populations as an evaluation of the ability of the model scenario output data to represent observed data.

Data and knowledge for each of the primary threats were reviewed and statistical analyses updated (see details in following sections). The results of the review and analyses were used to develop single and cumulative threat scenario models.

Model simulations were run on each scenario 10,000 times and summary statistics were recorded for population growth rate (*r*), population size at each time step (N_t), and probability of extinction (defined as only a single sex remaining). The population size at each year (mean and standard deviation) was compared to the observed (realised) population size for each population from the census survey data. Population growth rate (*r*) was quantified as the exponential rate of increase, according to the following equation:

$$r = ln\left[\frac{N_{t+1}}{N_t}\right]$$

In long-term simulations that reached an arbitrarily set maximum population size ("carrying capacity", K), the annual growth rate was calculated each year before the truncation of the population size to K, so that the *r* represented the intrinsic growth that would occur if a ceiling was not imposed on the population size. The model results (the predicted population size resulting from threat-modified reference vital rates) were then compared to the observed (realised) population dynamics from the census data over the same time period (2000–2017). The assumption of this approach is that if a model scenario replicates the realised dynamics for both the SRKW and NRKW populations then the model is appropriate for the system. The inspection approach method was used to validate the models (Law and Kelton, 1991); for the threat-modified model scenarios that most closely approached the observed population parameters, a simulation scenario with historical input data in place of the parameter randomly chosen from a distribution was also evaluated. In this case the yearly Chinook salmon index data was included in the historical scenario. A valid model should closely resemble the observed killer whale survey data when the historical data are used, including population size, age structure and sex ratio.

A sensitivity analysis was conducted on the prey, noise and contaminant parameters in the model to test the impact of uncertainty in these parameters on the results of the study. The sensitivity analysis was conducted in Vortex 10.3 using the Sensitivity Testing operations.

Once a model with acceptable performance was defined, model scenarios were projected into the future to examine the long-term population growth rate and future of the populations. The projection of the cumulative effects model from the 2017 population assumed that the current levels of threats continued into the future, with no changes in threats and no mitigation actions.

2.4. Threats

2.4.1. Prey availability

Field observation and statistical evidence support the relationship between the availability of Chinook salmon and mortality and fecundity rates for these populations (Ford et al., 2010; Ford et al., 1998; Vélez-Espino et al., 2015; Ward et al., 2009). Analysis of prey remains indicates that Chinook salmon can comprise up to 90% of the summer diet of SRKW (Ford and Ellis, 2006; Ford et al., 1998; Hanson et al., 2010).

Ford et al. (2010) reported that SRKW and NRKW survival rates were related to the modelled abundance of Chinook stocks available to six fisheries (Alaska Troll, BC North Troll, BC Central Troll, West Coast Vancouver Island Troll, Georgia Strait Sport, and Washington/Oregon Troll).

The statistical relationship between SRKW/NRKW mortality rates and Chinook salmon ocean abundance index values was updated using the entire time series of SRKW/NRKW and Chinook salmon data (1979–2017). The Chinook salmon stock index that best explains the mortality patterns seen in both populations was tested using model selection (Akaike Information Criterion, AIC). Linear regression was performed between SRKW/NRKW mortality with a one-year time lag and either the Coastwide Index (excluding Southeast Alaska (SEAK) stock, as done in Ford et al. (2000)), or the Chinook salmon runs deemed most relevant to each population (SRKW-stocks and NRKW-stocks). For SRKW-stocks, the WCVI + FL + OC runs were used (West Coast Vancouver Island, Fraser Late, and Oregon Coastal) and for NRKW-stocks the FE + PS + URB were used (Fraser, Puget Sound, and Upper Columbia River Brights) (Table A3) (Stredulinsky, 2016; Vélez-Espino et al., 2015).

To represent prey abundance in the models, Chinook salmon ocean abundance data were obtained from the DFO Salmon Program (A. Vélez-Espino, DFO, Pacific Biological Station) (1979–2017) from the 2018 Pacific Salmon Commission's (PSC) Chinook model calibration. Ocean abundance is an adequate representation of fish available for consumption by killer whales, given that the full time series of terminal run reconstruction data was unavailable, and ocean abundance has statistical support in previous analyses (Stredulinsky, 2016; Vélez-Espino et al., 2015). Yearly modelled ocean abundance was converted to an index of abundance by standardising the value by the mean for the full time series. The Chinook index value was randomly assigned in each model year using a normal distribution as defined by the median value (for a skewed distribution) and standard deviation from the entire time series. Selecting a value from a distribution in each year allowed the model to represent the fine temporal structure and variation in Chinook salmon abundance, and its impacts on killer whale vital rates. The distribution can then be used to project model scenarios over time periods not covered by the historical abundance data.

The availability of prey can also have significant effects on SRKW/NRKW reproductive success and the probability of calving. Ward et al. (2009) assessed calving probability (fecundity) of combined NRKW and SRKW females using a logistic regression model and found that fecundity was highly correlated with the PSC index of Chinook salmon abundance for the WCVI troll and recreational fishery in the prior year (one year lag). The model that best supported the data included age-structured effects on reproduction and a region effect to represent the lower calving rates in SRKW compared to NRKW. The logistic regression analysis was repeated with the additional 10 years of data for calving probabilities and PSC Chinook model ocean abundance salmon indices, following the statistical methods of Ward et al. (2009); additional details can be found in Appendix A. R code for the statistical analyses for prey abundance effects on mortality and fecundity can be found in Appendix B.

2.4.2. Vessel disturbance

Acoustic disturbance (noise) may come from a range of anthropogenic activities but this study focuses on the impacts of vessel-associated disturbance on killer whales. There is limited field evidence on the effects of vessel disturbance. Lusseau et al. (2009) observed a 25% reduction in SRKW feeding activity when boats were present. A noise exposure model combined with a Population Consequences of Disturbance (PCoD) model (National Research Council, 2005; Tollit et al., 2017) estimated that the lost foraging time for SRKW in the Salish Sea from a combination of behavioural responses and acoustic masking due to vessel presence was 20–23% of each whale-day (Tollit et al., 2017). Lacy et al. (2017) assumed that the effect on demographic rates of reduced feeding activity was the same as a comparable reduction in prey (i.e., no behavioural compensation by killer whales). In the PVA model, Lacy et al. (2017) estimated that vessels are present 85% of the daytime and killer whales are foraging in the presence of vessels 78% of the time. This represents a 16.6% reduction in Chinook salmon availability in the model (25% x 85% x 78%).

A time series of vessel activity for the study region that is comparable to the data available for killer whale population dynamics and Chinook salmon was not available. In order to estimate the relative presence of vessels in each population's range, data on the magnitude of vessel presence (commercial, recreational, and whale watching vessels) in the range of SRKW and NRKW were compiled (Appendix A). This estimate was used to set the vessel presence parameter for acoustic disturbance in the model. Acoustic disturbance was modelled as a reduction in feeding efficiency, and was directly linked to the variation in Chinook salmon abundance. The noise parameter was set to 0.85 for SRKW (equivalent to a 16.6% reduction in prey availability) and 1 for NRKW (no effect on prey availability).

2.4.3. Vessel strike

Fatal vessel strikes remove individuals completely from the population, affecting small populations disproportionately. Attributing cause of death in killer whales is difficult in many cases as carcasses often sink and are lost, meaning only a small proportion are recovered for necropsy examination (DFO, 2018; Ford et al., 1998; National Marine Fisheries Service, 2008). Limited data on cause of mortality suggest that SRKW have a slightly higher risk of strike than NRKW, 9.5% and 7.1% respectively between 1979 and 2017 (Appendix A; Baird, 2002; Ford et al., 2000; Williams and O'Hara, 2010). Changes in the frequency of vessel transits and the characteristics of ships (quieter or faster ships may increase strike risk) could affect this probability in the future. The vessel strike threat was modelled as an animal being removed randomly from the modelled adult population once every ten years. The probability was shared equally between males and females of the population.

2.4.4. Polychlorinated biphenyls contamination

The impact of environmental contaminants on killer whale vital rates was investigated using polychlorinated biphenyls (PCBs) as a representative class of contaminants due to a lack of available information on the impacts of other toxins. Hall et al. (2006, 2018) developed a PCB accumulation/depuration model to link PCB levels to calf mortality in cetaceans. The PCB model simulates the accumulation of PCBs in individuals over time, based on a set accumulation rate. Females offload an estimated 77% of their PCB loads to each calf during gestation and nursing (Hall et al., 2006, 2018). This PCB model has been used in cumulative effects assessment for SRKW (Lacy et al., 2017) and in estimating risk to global killer whale populations (Desforges et al., 2018). The dose-response logistic regression model curve used in these studies (Hall et al., 2018) was applied in the PVA scenario models to predict calf survival based on maternal PCB level. Contaminant model scenarios were run using either initial PCB levels from tissue samples summarised by Ross et al. (2000) (1993–1996) or the grand mean for the entire time series of tissue samples (Guy, 2018; Pearce and Gobas, 2018) (Appendix A; Table A10), with three different accumulation rates (1, 2, and 6 mg per year). The modelled PCB concentrations were then compared to the measured PCB levels in tissue samples.

3. Results

3.1. Pathways of Effects conceptual model

The overall PoE conceptual model (Fig. 2a) identified the important conceptual connections between threats (prey availability, acoustic and physical disturbance, and contaminants) and SRKW/NRKW population vital rates (fecundity, mortality), based on literature review and expert opinion. Prey availability appeared to be a central node, with six linkage

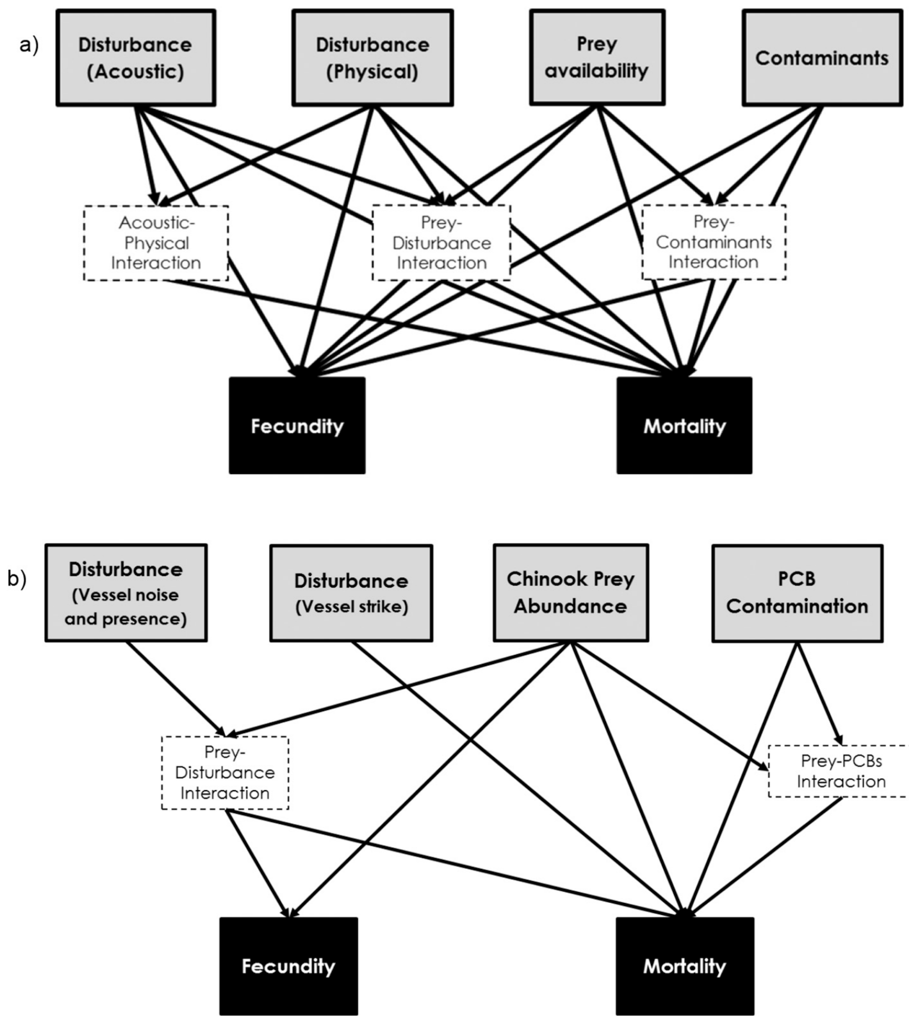

Fig. 2. a) Overall Resident Killer Whale Pathways of Effects conceptual model, including threats, interactions, and impacts on Resident Killer Whale fecundity and mortality and b) Population Viability Analysis model reduced to only the quantifiable threats and interactions.

pathways to fecundity and mortality, including two interactions with other threats (acoustic disturbance, physical disturbance) (Fig. 2a). Interactions make the assessment of impacts more difficult, as they imply that impacts may not be additive and instead may have non-linear or threshold effects. For example, the ability of killer whales to successfully catch and consume prey (access to prey) may be affected by disturbance. The impacts of disturbance could be exacerbated when prey abundance is low, and conversely, disturbance may have little or no effect on overall feeding efficiency when prey abundance is high (Prey-Disturbance Interaction). It has been hypothesised that killer whales might have a higher risk of vessel strike when exposed to loud sounds, which could impair the whales' ability to detect vessels and result in an acoustic-physical interaction effect (Erbe et al., 2018). The effects of PCBs on killer whales may be mediated by nutritional stress and the amount of blubber stores, as observed in seals, resulting in a prey availability-contaminants interaction (Robinson et al., 2018). Blubber-bound toxin levels are higher in Bigg's (transient) killer whales but their population is increasing rapidly (Ford et al., 2007) and therefore may not experience the same toxic effects as in prey-limited populations that are mobilising the toxins during periods of nutritional stress (Mongillo et al., 2016).

Based on the review of the available literature and data, only a subset of the linkages in the PoE conceptual model could be parameterised with empirical data and statistical relationships in the Population Viability Analysis (Fig. 2b). The Disturbance (acoustic) threat was represented by the combined effects of vessel noise and vessel presence as there was no way, with current knowledge, to separate impacts of vessel presence from those of vessel noise. The Disturbance (physical) threat was represented by the effects from vessel strikes. Prey availability was represented by Chinook salmon abundance in the PVA model, even though it is acknowledged that other types of salmon are also consumed. For the Contaminants threat, despite the evidence that other contaminants are present in killer whales, only PCBs could be included.

3.2. Population viability analysis

3.2.1. Threat scenarios

Scenarios for each of the individual threats were constructed and tested using the available knowledge and data (Appendix A). The baseline model using SARKW rates is shown in Fig. 3a. The best fitting statistical relationship between killer whale mortality and Chinook salmon abundance (1979–2017) included the relevant stocks for each killer whale population ($y = 1.6773–0.673\times$; $r^2 = 0.0889$, $p = 0.012$). The previous Ford et al. (2010) analysis used data up to 2003 and the addition of fourteen years of data reduced the explanatory power of the prey-mortality relationship. The best model to explain calving probability (lowest relative AIC value) included the relevant Chinook salmon stocks and an SRKW/NRKW age structure (Appendix A). The percentage of adult females breeding (Br) was defined as a logistic function with age structure, using separate parameters for younger (< 31 years of age; Br_1) and older females (> 30 years of age; Br_2). These coefficients were re-

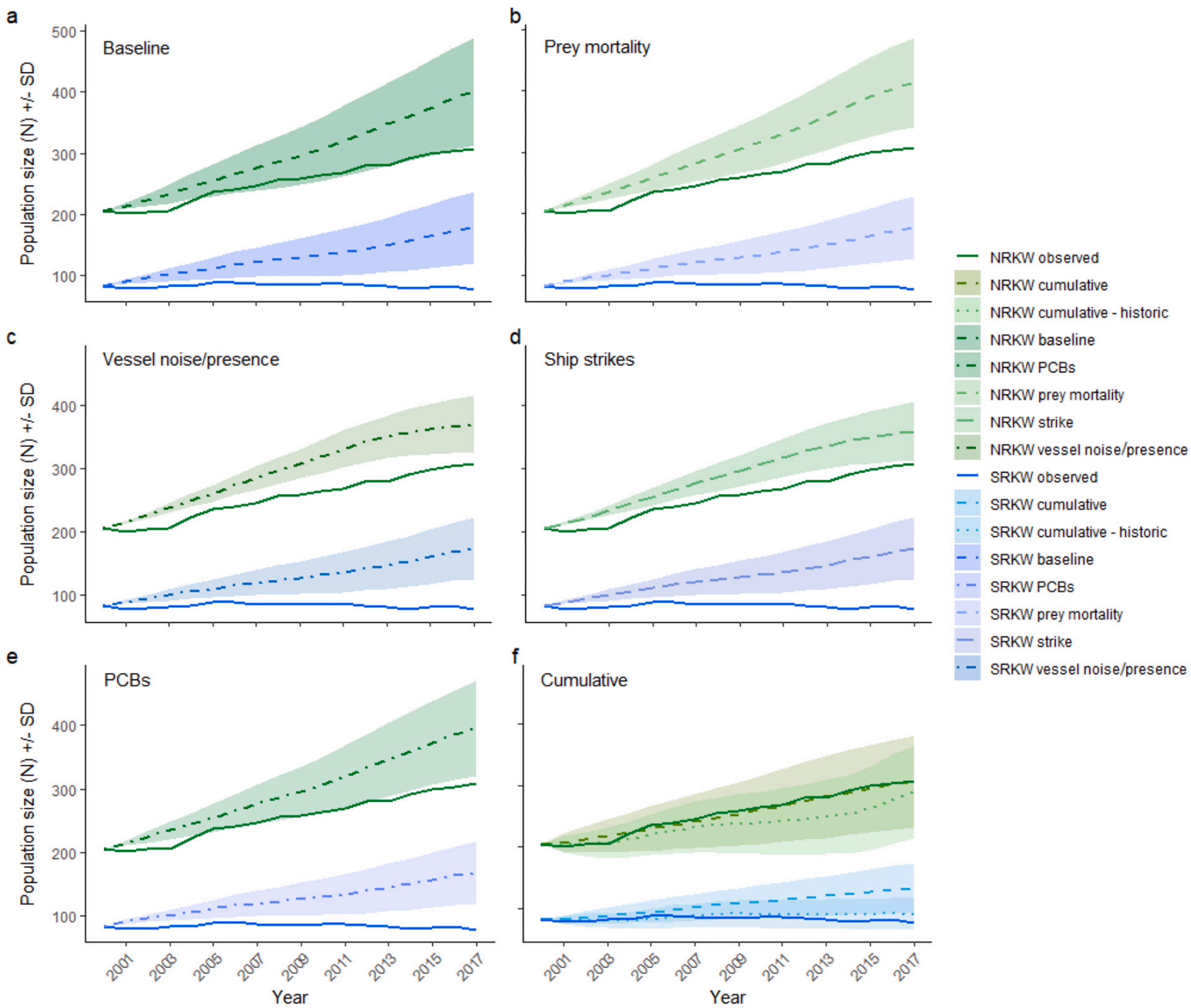

Fig. 3. Mean model simulations of population size (± standard deviation) for single threat scenarios a) baseline, b) prey mortality, c) vessel noise/presence, d) vessel strikes e) PCB contamination, and f) the cumulative effects and historic scenarios (all four threats) on NRKW (green dashed lines) and SRKW (blue dashed lines), with observed population size (solid lines). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

scaled for use in the model scenarios as a reduction to the reference fecundity rate.

In PVA scenario testing, prey abundance effects on mortality alone did not explain the realised population growth (Fig. 3b). The most realistic model for NRKW included the effects of prey abundance on both mortality and fecundity and approached the observed population trend, especially in the first 12 years of the simulation. For SRKW, the model scenario that incorporated impacts on both mortality and fecundity did not match the observed population trend, as it predicted slow population growth, rather than the observed decline. Additional combinations of Chinook stocks and distribution shape scenarios were tested but did not approach the observed population trends (details in Appendix A).

Vessel noise/presence scenarios tested the effect of noise as a reduction in prey abundance, with additional scenarios testing the model for the possibility of a threshold effect where noise affected feeding efficiency only when prey abundance was below the long-term average (Appendix A). None of the vessel noise/presence scenarios approached the observed population dynamics (Fig. 3c). The vessel strike threat scenarios did not match the observed population trends (Fig. 3d). These results suggest that the relatively rare vessel strike threat does not control the dynamics of these populations.

The PCB threat scenario simulations generated a range of mean PCB tissue concentrations in adults across different initial PCB levels and accumulation rates (Table A10). The model scenarios that most closely approached the range of measured PCB levels in recent SRKW/NRKW samples were those with initial PCB levels set to the grand means (SRKW: females = 17.5; NRKW: females = 4.9 mg kg-1 lw), with accumulation rates slightly higher in SRKW than NRKW (2 mg yr^{-1} and 1 mg yr^{-1}, respectively). However, the impact of PCBs alone did not match the observed population growth rate for either population (Fig. 3e).

3.2.2. *Cumulative effects*

The cumulative effects PVA scenario with all threats included (prey abundance, PCBs, vessel noise/presence and vessel strikes) was closer to the observed population sizes than any of the single threat models (Fig. 3f). The cumulative model included interactions between prey abundance and vessel noise/presence and PCB impacts, where the impact of vessel noise and PCBs was only applied when prey abundance was low (less than the long term mean index) (Table A11). The cumulative model approached the realised population growth for both populations closely, especially in the NRKW population (Fig. 3f). The mean model NRKW population size in 2017 was 309 (± 76 SD) individuals, the recorded NRKW population in 2017 was 308 individuals. The average model SRKW population size in 2017 was 134 (± 41) individuals, and the recorded SRKW population in 2017 was 77 individuals. Using the historical (rather than drawn randomly from the defined distribution) Chinook index values for 2000–2017 resulted in the cumulative effects model approaching the observed population growth even more closely, especially for SRKW (Fig. 3f). Historical values are a useful model validation, but the use of the distribution is needed in order to define a model that can be used for projection into the future.

To further validate the model, we compared the observed and simulated population structure in the cumulative effects model. The relative proportions of juveniles and adults were similar for both SRKW and NRKW (Table A12). The NRKW model was extremely close to the observed values in its outputs (Modelled: 102 juveniles and 207 adults; Observed: 104 juveniles and 204 adults). The observed sex ratios for both populations were also similar to those produced by the cumulative effects models, which both predicted more females than males.

3.3. Model projection

Model scenarios were projected into the future to examine the long-term population growth rate and future of the populations. The projection of the cumulative effects model assumed that the current levels of threats continue into the future, with no changes in threats and no mitigation actions. Future outcomes differed according to whether the full or a subset of the prey abundance time series was used as input to the model. When Chinook salmon abundance was randomly drawn from the long-term mean abundance distribution (1979–2017), the cumulative model projected mean positive population growth for both populations, but with uncertainty among iterations and across years that included negative population growth: 1.6% (± 7.9 SD) for NRKW and 1.5% (± 8.1 SD) for SRKW (Fig. 4). The NRKW population size reached the arbitrarily-set carrying capacity (500 individuals) early in the projections and this affected the projected future population sizes (which would have been higher in the absence of a set carrying capacity). The probability of extinction (defined in the model as only one sex remaining) for both populations was 0% over 100 years. In contrast, when the cumulative effects model used the recent (2008–2017) distribution of Chinook salmon abundance indices the model projected negative population growth for SRKW (−2.5% ± 10.5), and a slightly lowered, but still positive, growth rate for NRKW (Fig. 4). Under the prey scenario using the recent time series, SRKW had a 26.1% probability of extinction and in those simulations where extinction occurred, the mean time to extinction was 86 years (± 11.3 years).

3.4. Sensitivity

The sensitivity of model parameters was tested to distinguish which threats had the highest impact on long-term population dynamics within the model structure. Sensitivity testing was performed for the cumulative effects scenario model projection for SRKW, which includes all four threat variables as well as the defined interactions between disturbance, contamination, and prey availability. The parameter of interest was varied across its range (minimum-maximum) by set increments, with the base values used for all other parameters (Table 2). The base threat levels were the original values used in the cumulative effects model. For the prey parameter sensitivity testing, the full range of Chinook index values for both stocks (1979–2017) was tested (minimum = 0.4, maximum = 1.8). The vessel noise/presence parameter began at the base level of noise (0.85, equivalent to 16.6% reduction in feeding rate) and increased to a maximum of 1.55, to represent the possibilities that either the reduction in feeding time or the time vessels were present could be higher than estimated. Strike risk was varied from 5% to as high as 50%. The PCB value tested included the base initial PCB tissue concentration for females, and included the full range of measured female PCB tissue concentrations. Male PCB tissue concentration was not used in sensitivity testing because the impact pathway occurs via maternal transfer. The most sensitive parameter for the long-term projection of the population was prey abundance (the value of the Chinook index), followed by vessel noise/presence (Fig. 5). Lacy et al. (2017) previously conducted sensitivity analyses on the effect of demographic parameters on population growth in a similar SRKW PVA model and found that variation in fecundity had the strongest effect on population growth for this population.

4. Discussion

The cumulative effects assessment suggests that resident killer whale

Table 2
Parameters and values used for testing sensitivity in the SRKW cumulative effects scenario projection, including the base value, the range, and increment of testing.

Parameter	Base	Minimum	Maximum	Increment
Prey availability (Chinook Index value)	1.00	0.40	1.80	0.10
Disturbance (model value)	0.85	0.85	1.55	0.10
Female PCB tissue concentration (mg kg-1 lw)	17.46	5.00	200.00	25.00
Strike risk (probability)	0.10	0.05	0.50	0.05

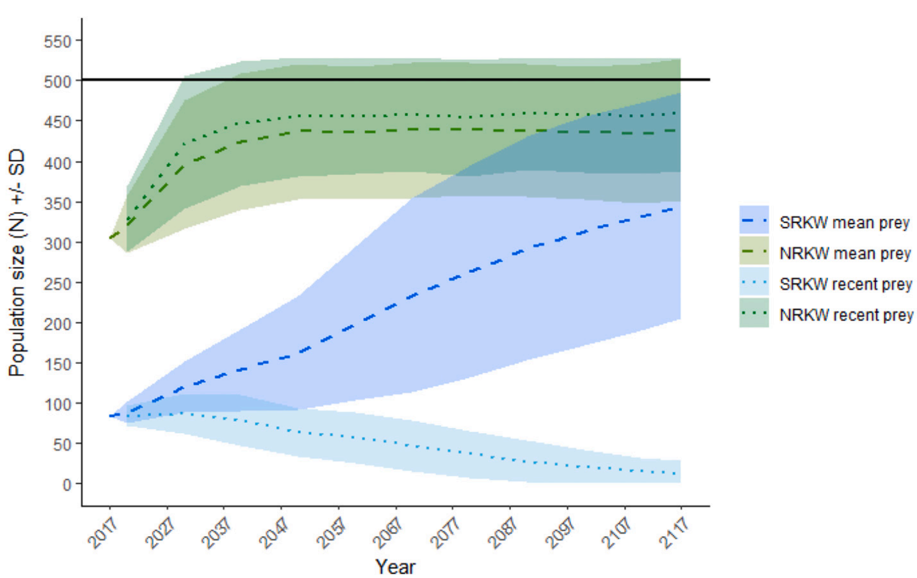

Fig. 4. Mean projection of the cumulative effects model 100 years into the future (starting in 2017) for NRKW (green) and SRKW (blue), under mean Chinook index ("mean prey": 1979–2017) or recent Chinook index ("recent prey": 2008–2017). Error bars represent ±1 standard deviation. Black horizontal line shows the arbitrary carrying capacity set for NRKW (500 individuals). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

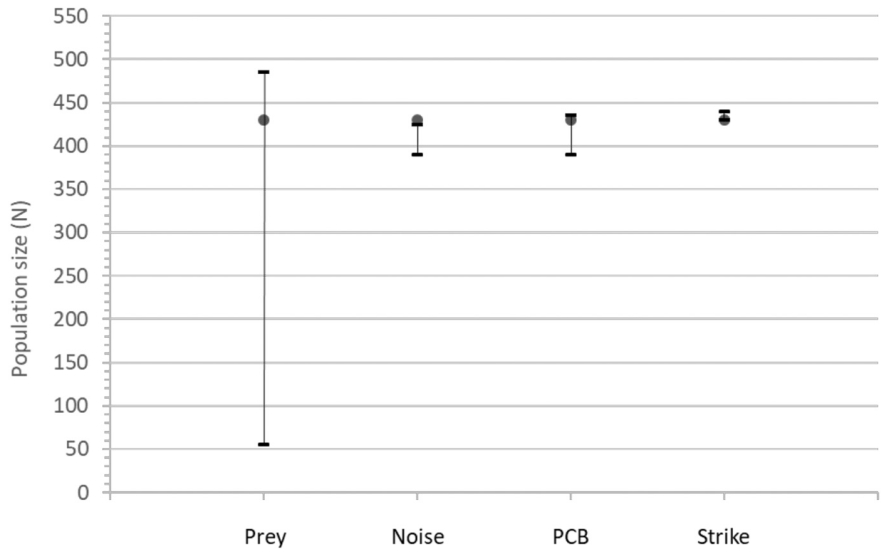

Fig. 5. Sensitivity of SRKW projected population size in the cumulative effects scenario to changes in the threat parameters: prey abundance, vessel noise/presence, PCB concentration and strike risk. Black circles represent the base value for each threat and the vertical bars represent the range of population size (N) with varying threat value (Table 1). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

populations are affected by the cumulative effect of anthropogenic threats and provides insights into the possible mechanisms behind the two populations' different trajectories. Chinook salmon abundance and its interactions with vessel noise/presence and PCB concentrations strongly influenced modelled killer whale population dynamics, highlighting the importance of considering threats collectively. The cumulative effects model fits the observed data, especially when recent low prey abundances were used, and is a more useful model than single threat models because it includes all threats and therefore can be used to examine tradeoffs in mitigation and management strategies. The cumulative effects PVA model can be used to project NRKW and SRKW population trajectories into the future. These projections are best used in a comparative evaluation of relative outcomes, rather than absolute predictions of abundance.

The projected population growth for both NRKW and SRKW was highly sensitive to the Chinook salmon abundance index. Under long-term mean Chinook salmon abundances, the modelled SRKW population was projected to increase; but when recent, lower Chinook salmon abundances were used, the SRKW population was projected to decline from present-day abundance, with a 26% chance of extinction within 100 years. Model projections were based on an assumption that modelled threat conditions will continue at the same levels as the present day with no mitigation. However, if the Chinook salmon stocks that SRKW depend upon continue to decline, this could affect the future outlook of the SRKW population, and potentially increase the probability of extinction above the model projection.

The findings of this cumulative effects assessment strongly support the significant role of prey availability in determining NRKW and SRKW population trajectories, and are consistent with previous work (Ford et al., 2010; Lacy et al., 2017; Vélez-Espino et al., 2015; Ward et al., 2009). The updated statistical analyses for the effect of prey availability on mortality and fecundity suggest that these impacts are still important to RKW, but the explanatory power of single-threat models has been reduced compared to previous studies. Sensitivity testing showed that prey abundance had the greatest effect on model results, within the bounds of the model structure. Improved mechanistic understanding of the other threats is still needed and could be used to advance the model for projections into the future. The cumulative effects model employed interactions of prey abundance with both vessel disturbance and contaminants, but these mechanisms have not been validated. The most uncertainty among the threats is related to the impacts of underwater noise and vessel disturbance. Additional research is urgently needed on the impacts of vessel presence and noise disturbance on resident killer whales. There are no comparable time series for vessel traffic and proximity to killer whales that would allow similar statistical testing to that done for prey availability. The current work contained a mechanistic model of PCB contamination but other contaminants are also a concern. PBDEs have also been found in high concentrations in these populations (Ross, 2006), although there was insufficient data available to include in the model. An important assumption made in this work is that the pathways of effects from threats to impacts are the same for both SRKW and NRKW; in other words, that the mechanisms by which threats affect individuals are the same for both populations. This assumption is the justification for using the same impact model structure for both populations, albeit with differing threat levels. The consequences of exposure to threats are assumed to be the same for both populations, while the level of exposure to threats is assumed to be population-specific. Differences in distribution, genetics, behaviour and other ecological characteristics at the sub-population (pod/clan) level may affect the exposure to threats and these nuances were not captured in the current assessment. The relationships between threats and resident killer whale mortality and fecundity were determined based on knowledge mostly obtained in the Salish Sea area in the summer/fall period but were assumed to be representative of relationships throughout the entire NRKW and SRKW ranges over the entire year.

Further, the two populations may exploit different prey stocks that themselves have varying population dynamics and availability to killer whale predation. All Chinook salmon stocks went through a period of decline in the 1990s, but since then have experienced stock-specific temporal variation (Ford et al., 2010). The ability and flexibility of killer whale populations to exploit different Chinook salmon stocks, other salmon species and indeed other fish taxa is not fully understood and may vary between NRKW and SRKW and through time. Potential prey competition between the two killer whale populations, and with other marine mammals, such as pinnipeds, may also affect prey availability and has not been included in the current models.

The positive population growth projected by the cumulative effects model under mean prey abundance assumes that the current levels of threats will not increase from present levels, which may not be the case in reality. Changing climate conditions and an increasing human population are having significant ongoing impacts on the marine environment and are likely to continue to affect killer whales and their prey into the future (DFO, 2018; Harley et al., 2006; Walsh et al., 2020). Reductions in threats, such as through mitigation and management actions, may also improve future prospects for positive population trajectories. The USA and Canada have taken a number of management actions in recent years to support the recovery of the SRKW population, including Chinook commercial and recreational fishery closures in key killer whale feeding areas (DFO, 2018). Incorporating the effects of management actions, changing natural conditions, and changes to threat levels into iterations of the cumulative effects assessment may provide useful insights into the potential impacts of these actions on projected population trajectories.

Threats with low probability and high population consequences, such as oil spills or disease outbreaks, are difficult to include in simulation modelling. These threats should not be ignored in management and mitigation because they can have catastrophic consequences if the population were to be exposed. The *Exxon Valdez* oil spill in Alaska was linked to a significant decline in a resident killer whale pod (AB) that had been observed swimming through spilled oil; this pod suffered significant losses in the year following the spill, and had still not recovered to pre-spill levels 16 years after the event (Matkin et al., 1999; Matkin et al., 2008). One way to address high consequence, low probability events such as oil spills in model simulations could be to dramatically reduce the population to 50–75% of the current levels and test if the model population would be resilient enough to recover from such a catastrophe.

This cumulative effects assessment further advances the field by combining a detailed Pathways of Effects conceptual model with a Population Viability Analysis simulation model (after Lacy et al., 2017) to evaluate how the current state of human activities might affect the future persistence of the two imperilled killer whale populations. The incorporation of a PoE model allows the inputs and structure that inform the quantitative PVA to be explicit, and identifies areas lacking knowledge that were not able to be included but could be of value in future iterations. The cumulative effects PVA model could be a useful tool for testing the potential impacts of different theoretical mitigation and management scenarios for individual threats on population trajectories; for example to test whether the complete mitigation of acoustic disturbance would cause the projected population trajectory to increase over time and how long it may take for a change in population trajectory to be observable. Different parameters (e.g., increased vessel presence) can be input into the cumulative effects PVA model to consider the potential impacts of proposed developments and other anthropogenic changes. New information from ongoing and/or planned further research such as prey competition in key foraging areas, foraging efficiency, diet composition, prey field analysis, underwater acoustic monitoring and modelling, and contaminant sources and levels, will all help to inform future iterations of the PoE and PVA models. These models can help to adaptively inform and/or implement recovery measures, such as investigating the benefits of management actions to protect important areas, evaluating potential impacts of disturbance and prey competition from fisheries, assessing the potential impacts of salmon enhancement, and assessing industrial project impacts on killer whales and their habitat to provide advice on avoidance and mitigation measures. Population viability models have been used in conservation biology for over 30 years (Lacy, 2018) with many different approaches. The cumulative effects assessment case study described here builds on this considerable knowledge base and can provide guidance for assessments in other imperilled species.

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CRediT authorship contribution statement

Cathryn Clarke Murray: Conceptualization, Methodology, Formal Analysis, Investigation, Writing-Original draft preparation, Writing - Review and Editing, Supervision. **Lucie Hannah**: Conceptualization, Methodology, Investigation, Writing – Original Draft Preparation, Writing - Review and Editing. **Thomas Doniol-Valcroze**: Conceptualization, Writing – Review and Editing, Supervision. **Brianna Wright**: Investigation, Formal analysis, Writing – Review and Editing. **Eva Stredulinsky**: Investigation, Formal analysis, Writing – Review and Editing. **Jocelyn C. Nelson**: Formal analysis, Visualization, Writing – Review and Editing. **Andrea Locke**: Conceptualization, Writing – Review and Editing. **Robert C. Lacy**: Methodology, Software, Writing – Review and Editing, Supervision.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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UNITED STATES DISTRICT COURT
WESTERN DISTRICT OF WASHINGTON
AT SEATTLE

WILD FISH CONSERVANCY NORTHWEST, a Washington non-profit corporation, Plaintiff, v. BARRY THOM, in his official capacity as Regional Administrator of the National Marine Fisheries Service, *et al*, Defendants.	Case No. 2:20-cv-00417-MLP **DECLARATION OF DR. ROBERT LACY, Ph.D.**

I, Robert Lacy, state and declare as follows;

1. I am over eighteen years of age. I have personal knowledge of the facts contained in this declaration and am otherwise competent to testify to the matters in this declaration.

2. I received my B.A. and M.A. in Biology from Wesleyan University in 1977, where I graduated summa cum laude. I received my Ph.D. in Evolutionary Biology with minors in Genetics and Ecology from Cornell University in 1982. I serve on the faculty of the Committee on Evolutionary Biology at University of Chicago. I was a Conservation Scientist for the Chicago Zoological Society from 1985, until my recent retirement and appointment as a Conservation Scientist Emeritus. Although “retired” I still work actively with the Species

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Conservation Toolkit Initiative, a team that develops, distributes, and supports software for species risk assessments and wildlife population management.

3. My qualifications, including publications, is contained in my Curriculum Vitae, which is attached as Exhibit B to this declaration.

4. I have been retained by Wild Fish Conservancy, through its counsel, to provide expert opinions in this matter on issues related to the Southern Resident Killer Whale population and the implications of the National Marine Fisheries Service's ("NMFS") conclusions in the Biological Opinion issued with regard to the 2019 Pacific Salmon Treaty. This declaration describes my opinions and the bases therefor.

5. In addition to drawing upon my knowledge and expertise, I have reviewed the materials cited throughout this declaration and those identified in the list of cited materials attached to this declaration as Exhibit A in developing my opinions expressed herein.

6. In summary, the opinions I express herein are as follows:

a. Analyses conducted in 2015 projected that the Southern Resident Killer Whale population would decline slowly at a rate of about 0.2% per year if environmental conditions and the demographic responses to threats remained as they had been over the previous few decades. Updated analyses on the current population now project about a 1% annual decline, leading to eventual extinction of the population as demographic and genetic problems become worse with the ongoing decline in the breeding population. The numbers of Southern Resident Killer Whales increased from 1976 to a peak in 1993-1996, and has subsequently declined. The 2015 prediction of approximately zero population growth accurately reflected the lack of growth in numbers over the entire time period

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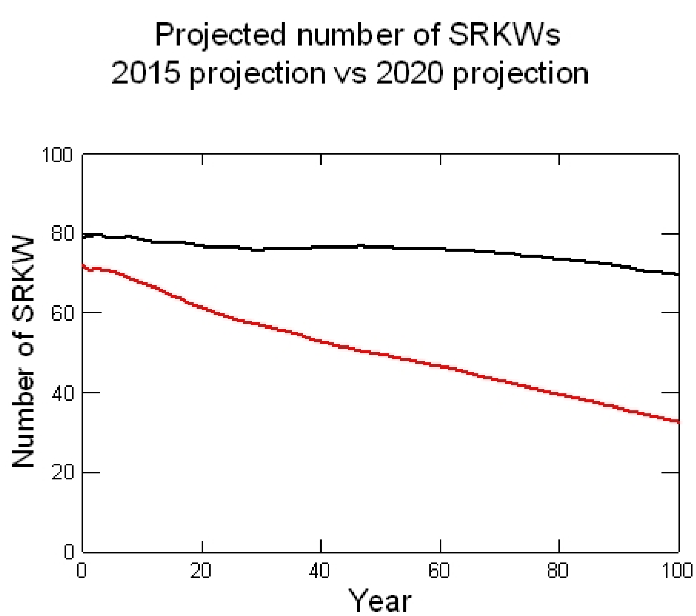
from 1976 to 2020, while the more pessimistic current prediction accurately mirrors the 1% average annual decline that has occurred since 1993. Since 2014, the Southern Resident Killer Whale population has declined at an even faster rate of about 2% per year. Although the difference between a 0.2% annual decline and a 1% annual decline might not seem large, the cumulative effect of the faster rate of decline compounds to become considerable damage across the years. The following graph shows the mean projected number of Southern Resident Killer Whales, using the data from 2015 (upper, black line) and the mean projected number using the current (2020) data (lower, red line). In 2015, we estimated a 9% probability that the population would become functionally extinct with fewer than 30 animals within the next 100 years. With updates to reflect the current situation, I now estimate a 59% probability that the population will drop below 30 animals sometime in the next 100 years, becoming functionally extinct.



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b. The abundance of Chinook prey influences the reproductive rate and the survival rates of the Southern Resident Killer Whale. Analyses indicate that prey abundance is the factor that has the largest impact on Southern Resident Killer Whale population growth or decline. Using published estimates of the effect of prey abundance on demographic rates, we calculate that Chinook total abundance available as prey to the Southern Resident Killer Whale needs to increase by about 10% over the mean levels of the last few decades for the decline of the Southern Resident Killer Whale to be halted. Recovery of the Southern Resident Killer Whale population at the rate (2.3% growth) specified for delisting in the species' Recovery Plan will require an increase in the Chinook prey abundance of about 35%.

c. The NMFS 2019 Biological Opinion ("2019 SEAK BiOp") proposes several actions aimed at increasing the number of Chinook salmon available to the Southern Resident Killer Whales. The reduction in the Southeast Alaska salmon fishery of up to 7.5% in the 2019 Pacific Salmon Treaty relative to the preceding agreement, which is described in the 2019 SEAK BiOp, results in very little change in the Chinook available to the Southern Resident Killer Whales, and therefore would not have a measurable benefit for the endangered Southern Resident Killer Whale.

d. A proposed hatchery expansion aims to increase Chinook available to the Southern Resident Killer Whales by 4-5%. That increase in prey can be estimated to reduce the annual rate of decline of the Southern Resident Killer Whale population from about 1% to about 0.5%, but this would not be sufficient to stop

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the slide toward extinction.

e. The benefits to the Southern Resident Killer Whales of other possible mitigation measures are not quantified in the 2019 SEAK BiOp, and those actions would need to amount to a further increase (above that achieved from the two above mentioned measures) of at least another 5% in the Chinook abundance available as prey to Southern Resident Killer Whales in order for me to predict that the decline of Southern Resident Killer Whales would stop.

f. More aggressive management actions would be required to start the Southern Resident Killer Whale population on a reasonably secure path toward recovery or to meet NMFS' annual population growth rate goal of 2.3%.

7. My career has focused on building the capacity of the world to be much more effective in ensuring the long-term sustainability of species. I have done this via advancing the basic science that must underlie successful programs for sustaining species; providing the accessible tools to enable others to apply the science to species assessments, conservation planning, and population management; training students and colleagues in the use of the tools; and – when necessary – doing the analyses that inform and guide conservation for individual species.

8. Over my career I have developed, freely distributed, and supported software tools for guiding species conservation and population management. My approach has always been to provide tools for powerful and flexible analyses, within user interfaces that are accessible to wildlife managers, students, and others who might not have expertise with computer languages and systems. Consequently, the tools are now used globally to guide population management in nature reserves and zoos, viability analyses and recovery planning by wildlife agencies, and

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integrated assessment of threats to species. The software is used also to teach students about population biology and conservation in many universities.

Population Viability Analysis

9. Population viability analysis (PVA) is a class of scientific techniques that uses demographic modeling to assess risks to wildlife populations and evaluate the likely efficacy of protection, recovery, or restoration options (Shaffer 1990; Boyce 1992; Burgman et al. 1993; Sjögren-Gulve and Ebenhard 2000; Beissinger and McCullough 2002; Morris and Doak 2002). (All references cited in this Declaration are listed in Exhibit A.) PVA usually starts with standard demographic analysis ("life table analysis") to make deterministic projections of the expected population growth rate from the mean birth and death rates (Ricklefs 1990; Caswell 2001). PVA then extends the standard demographic projections in two important ways: (1) the impacts of forces external to the population (e.g., changing habitat quality, extent, and configuration; interactions with other species in the community; impacts of disease or contaminants; harvest, incidental killing, or other direct human impacts) on the demographic rates are explicitly considered and evaluated, and (2) uncertainty in the population trajectory caused by intrinsic (e.g., demographic stochasticity, limitations in local mate availability or other density dependent feedbacks, inbreeding impacts) and extrinsic (e.g., environmental variation, occasional catastrophes) factors can be explicitly modeled, usually through the use of simulation modeling. The outputs of PVA include any desired measure of population performance, but commonly assessed metrics include projected mean population size (N) over time, population growth rates (r), expected annual fluctuations in both N and r, probability of population extinction, and probabilities of quasi-extinction (the likelihood of N falling below any specified number within a specific number of years). These outputs are used to assess risk (e.g., for listing under the

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Endangered Species Act or other protective regulations), assess vulnerability to possible threats, determine sustainable harvest in the context of uncertainty, and determine the suites of actions that would be needed to achieve stated resource protection or restoration goals.

10. A requirement for any PVA model to provide sufficiently accurate and robust projections to allow estimation of population performance is the availability of detailed demographic data. Model input is required from the focal population or comparable reference populations for mortality rates, aspects of reproduction (e.g., age of breeding, age of reproductive senescence, inter-birth intervals, and infant survival), population size, and habitat carrying capacity – as well as the natural fluctuations in these rates. The difficulty in obtaining sufficient demographic data on endangered or protected species is a common challenge to the usefulness of PVA models, and many practitioners consequently recommend that PVA models be used only to provide assessments of relative risk and relative value of management options, rather than absolute measures of population trajectories. In the case of the Southern Resident Killer Whale population, however, demographic data are available from studies by the Center for Whale Research and others that are unprecedented in duration and detail of data collection. This exceptional data set provides a complete census of the total abundance as well as the age and sex composition of the Southern Resident Killer Whale population from 1976 to 2020. This allows for much more accurate projections of population performance and the ability to compare predicted trajectories to the precisely documented fate of the population.

11. PVA models were developed initially for quantifying future risk to populations that are vulnerable to collapse due to a combination of threatening processes (Shaffer 1990). They were soon recognized to be more reliable for assessing relative risk than absolute probabilities of decline or extinction (Beissinger and McCullough 2002; but see Brook et al.

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2000 for evidence that even absolute predictions of population trends can be accurate), and have become most useful in the identification of conservation actions that are most likely to achieve conservation goals (Sjögren-Gulve and Ebenhard 2000). The same methods can be used to quantify injury caused by an externally imposed stress, by comparing measures of population performance in the presence vs. absence of the stress, and to determine what actions would be needed to reverse the impact, restore the population to pre-injury health, and compensate for interim losses. The PVA forecasts can then be used to set the targets for expected performance under proposed restoration plans.

12. The Vortex PVA model that I developed (Lacy and Pollak 2020) is what is known as an individual-based model that projects the fate of each individual in a population. It simulates the effects of both deterministic forces and demographic, environmental and genetic stochastic (or random) events on wildlife populations. Vortex models population dynamics as sequential events that are determined for each individual in a population with probabilities determined from user-specified distributions. Vortex simulates a population by stepping through a series of events that describe an annual cycle of a sexually reproducing organism: mate selection, reproduction, mortality, dispersal, incrementing of age by one year, any managed removals from, or supplementation to, the populations, and limitation of the total population size (habitat "carrying capacity"). The simulations are iterated to generate the distribution of fates that the population might experience. Vortex tracks the sex, age, and parentage of each individual in the population as demographic events (birth, sex determination, mating, dispersal, and death) are simulated. A detailed description of the program structure is provided in Lacy (1993; 2000) and details about the use of Vortex are provided in the manual (Lacy et al. 2020).

13. The Vortex PVA modeling software is well-suited for the analyses of threats to

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the Southern Resident Killer Whale population, as Vortex is the most widely used, tested, and validated individual-based PVA model, and it is publicly accessible so that anyone can re-examine and repeat published analyses. It is highly flexible in allowing all input demographic parameters to be specified optionally as functions of external forces or as rates that change over time. Vortex has been used for modeling population dynamics of various marine mammal species (including bottlenose dolphins, Indo-Pacific bottlenose dolphins, baiji, manatees, dugongs, Hawaiian monk seals, and Mediterranean monk seals), as well as thousands of other species. Vortex has been shown to produce projections that accurately forecast dynamics of well-studied populations (Brook et al. 2000). Both NMFS in its 2019 SEAK BiOp (e.g., pp. 86, 90, 311) and Fisheries and Oceans Canada (Murray et al. 2019, e.g., pp. 3-5, 30, 33, 44, 62) have relied on analyses completed with Vortex for assessing the status of the Southern Resident Killer Whales.

Southern Resident Killer Whales

14. In 2015, at the request of Canada's National Energy Board ("NEB"), I led a team of six scientists conducting a PVA of the risk associated with aspects of the proposed Trans Mountain Expansion Project (Project) on the endangered Southern Resident Killer Whales. In that analysis, the PVA model was used to estimate the increased risk to the Southern Resident Killer Whales from three threats associated with the marine shipping component of the Project: an oil spill, increased acoustic and physical disturbance from ships, and ship strikes. The report also examined the possible effects of decreased Chinook salmon prey base that might result from climate change or human activities, and evaluated those impacts in comparison to the more immediate threats of the proposed Project and as the environmental context within which the impacts of the Project are likely to occur. The report to NEB (Lacy et al. 2015), including

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detailed descriptions of the methods and the data used in the PVA, is publicly available at http://docs.neb-one.gc.ca/fetch.asp?language=E&ID=A4L9G2. The analyses were extended and published in a peer-reviewed scientific paper (Lacy et al. 2017). Further updating of analyses using demographic data on the population through 2018 (Lacy et al. 2018) was submitted to NEB and is available at https://apps.cer-rec.gc.ca/REGDOCS/Search?txthl=A96429-3%20A%20-%20Expert%20Report%20of%20Lacy%20et%20al%20-%202018%20-%20Final%20-%20A6L5R2.

15. As of 2015 and 2017, based on status quo conditions, we projected the Southern Resident Killer Whale population would remain about at its current size or continue a very slow decline (estimated at a mean annual decline of 0.2%). We projected a 9% chance of quasi-extinction within the next 100 years, where the population falls below 30 whales and is no longer viable.

16. I have now updated the PVA model again, using fecundity and survival rates calculated from the detailed records from 1976 through 2018 and applying those rates to the current population of 72 Southern Resident Killer Whales. The following graph shows the mean projected population size (heavier, middle line) and the uncertainty in the trajectory (upper and lower lines showing ± 1 standard deviation among independent repeated simulations of the population).

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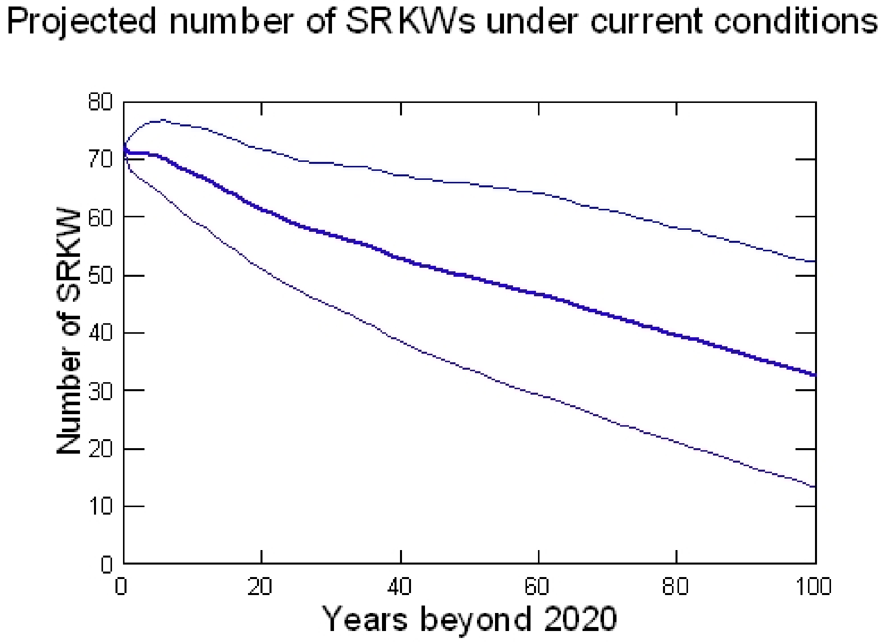

17. With current data, and if the Chinook availability remains at the mean level of the past few decades, the model projects a mean annual decline in the population of Southern Resident Killer Whales of about 1.0%. This is close to what has been occurring recently, and it compares to our 2018 projection of a smaller decline of 0.6% per year (Lacy et al. 2018). About half of difference between the 2018 and 2020 projections is due to the fact that the population is aging (with the mean age of living whales now just over 22 years, whereas it was just over 21 years in 2018), and more animals are now post-reproductive or nearing post-reproductive age. The other half of the difference is due to the fact that we now have parentage data for more of the animals, and that allows us to have more complete estimates of kinships among animals, and that in turn leads to slightly higher estimates of current and future inbreeding.

18. For our model, we obtained estimates of the impact of Chinook prey abundance on the reproductive rates and survival rates of the Southern Resident Killer Whales from published scientific reports (Ward et al. 2009; Velez-Espino et al. 2015; Ford et al. 2010). We

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scaled the numerical relationships so that the mean demographic rates observed in the Southern Resident Killer Whales from 1976 through 2015 were correctly predicted. (The details of the methodology are documented in Lacy et al. 2015 and Lacy et al. 2017 publications.) We then use these relationships to project the Southern Resident Killer Whale population trajectory in several scenarios that tested the impact of prey availability, expressed as a percent change in the annual abundance of Chinook salmon available as prey to the Southern Resident Killer Whales from the mean level over the last three decades.

19. The abundance of Chinook varies over time, and that variation in prey can be entered into the PVA model. However, as documented in the 2019 SEAK BiOp, the extent of that variation is very dependent on which stocks of Chinook are assessed, and it is not known precisely what proportion of the Southern Resident Killer Whale diet is composed of salmon from each stock. I examined the model projections with the Chinook abundance varying randomly across years around the long-term mean values being tested. I found that such an elaboration of the model had very little effect on the long-term projections for the Southern Resident Killer Whale population. This occurs because killer whales are very long-lived and slow breeders, so year to year fluctuations in demography will average out over their lifespans. Therefore, as was done in our prior PVA reports, the results from analyses presented in this declaration assume that the abundance of Chinook is at a fixed level each year and does not vary randomly around that value.

20. Also included in the model are the current estimates of both PCBs and noise disturbance, based on published estimates of the current magnitudes and effects of these threats (Hall et al. 2011; Hall and Williams 2015; Lusseau et al. 2009). These threats are part of the current environment for the Southern Resident Killer Whale, and they interact with the effect of

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prey limitation. (The documented impact of noise disturbance is via a reduction in time that the Southern Resident Killer Whales spend feeding. The primary impact of PCBs is on survival of calves, compounding the reduction in survival that occurs with low prey availability.) Only with these effects of PCB and noise disturbance in the model do we accurately predict the recent observed rate of decline of the population. However, even if these other threats were completely eliminated—which is not possible in the near term and unlikely in the long term—our modeling shows that there would not be adequate prey available to achieve the population growth goal established in the Recovery Plan for the Southern Resident Killer Whale (Lacy et al. 2017).

21. By applying the published relationships of Southern Resident Killer Whale reproductive and survival rates to Chinook abundance, and then testing the benefits to Southern Resident Killer Whales of incremental improvements in the abundance of Chinook prey, the model shows that to achieve a mean zero population growth (i.e., to stop the decline), there would need to be a sustained 10% increase (relative to the 1976-2015 average) in the mean abundance of the Chinook stocks available as prey to the Southern Resident Killer Whales.

22. The analyses conducted in 2015, 2017, and 2018 estimated that a 30% increase in Chinook could achieve the 2.3% growth called for in the Southern Resident Killer Whale Recovery Plan. With the further decline that has occurred in the population in the last few years, our analysis of the 2020 population now projects that a 30% increase in Chinook would result in about 2% growth per year, and a 35% increase in prey would be necessary to meet the recovery goal. The graph below shows the expected Southern Resident Killer Whale population growth across a range of levels of Chinook abundance. The two horizontal lines indicate zero population growth and the 2.3% growth goal of the Recovery Plan.

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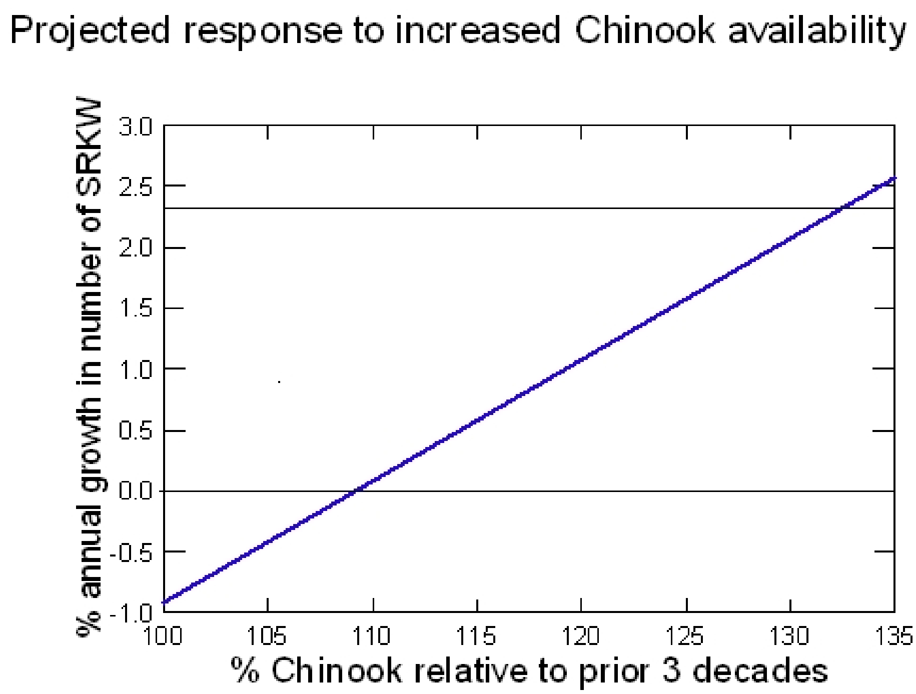

NMFS' Biological Opinion and Impact on Southern Resident Killer Whale Population

23. I was provided with NMFS' 2019 SEAK BiOp for Southeast Alaska salmon fisheries at issue in this matter. I reviewed it closely. In the 2019 SEAK BiOp, NMFS acknowledges that the Southern Resident Killer Whale population is declining, and that is at least partly and maybe mostly due to inadequate prey availability. The 2019 SEAK BiOp cites my previous work (p. 311) as evidence that the biggest threat is that lack of prey, although other factors such as noise, PCBs, oil spills, and other environmental factors all make things worse.

24. In several places, and in various ways, the 2019 SEAK BiOp estimates the reduction in prey available for Southern Resident Killer Whales caused by the Southeast Alaska fisheries (e.g., Tables 41, 42, and 97) as between 2-15% in coastal fisheries and 1-2% in inland fisheries. However, there is significant uncertainty depending on which salmon stocks and for which years the calculations are based. Importantly, the BiOp does not explain how the various percentage reductions mentioned translate to corresponding changes in the total mean abundance of Chinook that provide potential prey for Southern Resident Killer Whales, which is what is

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required for accurate projections of the benefits expected from reductions in the fisheries. The 2019 SEAK BiOp directly states (p. 94) "the impact of reduced Chinook salmon harvest on future availability of Chinook salmon to the Southern Residents is not clear."

25. The 2019 SEAK BiOp also discusses possible mitigation measures, which could increase the prey availability for Southern Resident Killer Whales. The 2019 SEAK BiOp estimates the newly negotiated 2019 Pacific Salmon Treaty will reduce the Southeast Alaska fishery annual harvest of Chinook by up to 7.5% relative to the harvest under the 2009 Treaty. A proposed increase in hatchery production mitigation seeks to provide 4 to 5% increase in prey available to the Southern Resident Killer Whales. The increase in hatchery production is not yet funded, so I would expect a delay of at least 5 to 10 years to account for allocation of funds, construction of any new facilities, increased programs of production, and then return of hatchery raised Chinook as mature adults.

26. I applied these estimates from the 2019 SEAK BiOp to the Vortex PVA model, in order to project the consequences of the possible scenarios described in the 2019 SEAK BiOp. The estimated 7.5% (maximum) reduction in the Southeast Alaska fishery, applied to a typical 6% reduction in prey available to the Southern Resident Killer Whales caused by the Southeast Alaska fishery as a whole (the 6% being an approximate middle value from the many estimates made in the BiOp), results in a less than 0.5% increase in the Southern Resident Killer Whale prey. This is only 1/20th of the 10% increase that is needed to achieve even a cessation of the decline in Southern Resident Killer Whale population.

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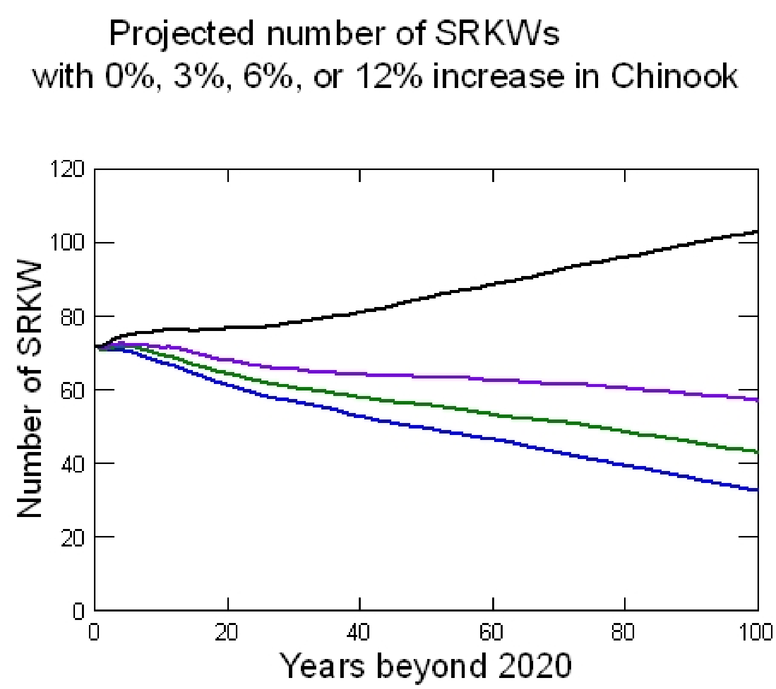
27. To estimate the possible reductions in threats to the Southern Resident Killer Whale that might be achieved with greater reductions in the Chinook fisheries, I projected a Southern Resident Killer Whale population growth with an immediate 6% increase in Chinook prey, and a 3% and a 12% increase in prey (half and double the middle estimate, covering most of the range of values reported in the 2019 SEAK BiOp for specific stocks and years). As shown in the following graph, with the existing baseline in blue (bottom line), the PVA projections for these scenarios show that the 3% increase in Chinook results in a mean 0.7% decline in Southern Resident Killer Whale population per year (green line), the 6% increase in Chinook results in a mean 0.4% decline of the Southern Resident Killer Whale population (purple line), and the 12% increase results in 0.3% positive growth annually (top, black line).



28. The impacts on Southern Resident Killer Whales of other estimates of prey increases that could be achieved by reductions in the fisheries can be extrapolated from the projections of Southern Resident Killer Whale population growth across a range of levels of Chinook abundance, as shown in the graph in paragraph 22, above.

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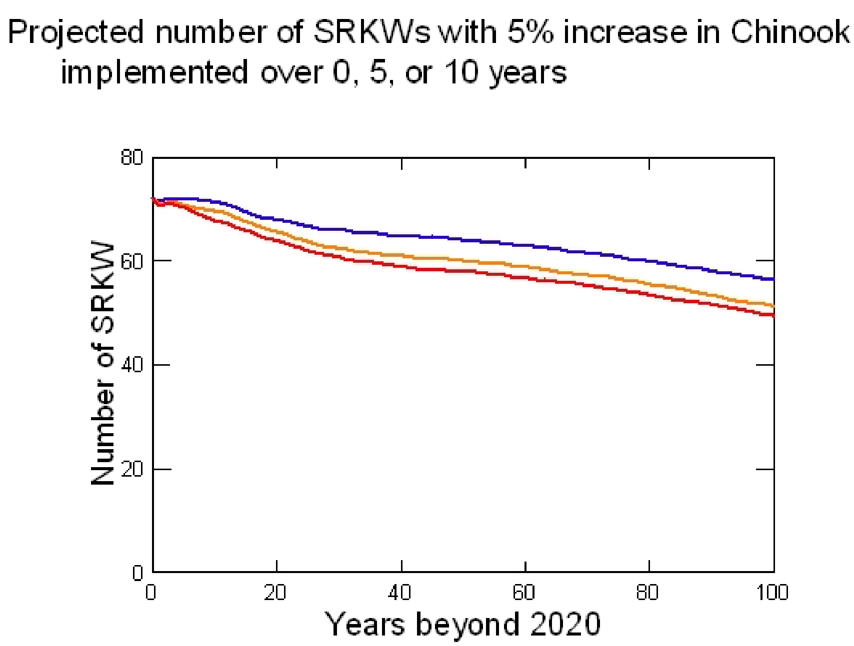
29. I projected the benefits to the Southern Residents of possible (but not yet funded) hatchery projects assuming a 5% increase in Chinook, beginning either 5 years or 10 years in the future. With either time scale for implementation and return of the hatchery-produced Chinook, the mean long-term consequence is a slowing of the decline in Southern Resident Killer Whales from 1.0% to 0.5% per year; therefore, not enough improvement to completely halt the decline. The difference between a 5-year delay and a 10-year delay in enhancement is that by year 10, the slower implementation will result in the Southern Resident Killer Whale population having declined by about 2 more whales before the improvement can begin to take effect. The following graph shows the projections if the mitigation measures achieve a 5% increase in Chinook (as estimated from the proposed hatchery expansion) instantly (top, blue line), after 5 years (middle, orange line), or after 10 years (bottom, red line). As this graph plainly demonstrates, delays in implementation of these theoretical mitigation measures have a very real and lasting impact on the Southern Resident population. Notably, it also shows that the proposed measure – even if implemented immediately – is not enough to stop the decline of Southern Residents.



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30. Combining the actions of reducing the Southeast Alaska Chinook fishery and increasing abundance to the Southern Resident Killer Whale of hatchery-raised Chinook, and possibly other mitigating actions as well (such as additional reductions in additional fisheries managed under the Pacific Salmon Treaty), could achieve the 10% increase in prey necessary for stabilization of the Southern Resident Killer Whale population or even greater increases in prey that would allow for recovery of the Southern Resident Killer Whales. Importantly, however, none of the scenarios proposed in the 2019 SEAK BiOp are projected to achieve this 10% increase in prey abundance. The analyses described above in paragraph 22 document the long-term growth in the Southern Resident Killer Whale population that could be achieved if Chinook abundance is increased by 35% above the mean levels of the last three decades.

31. Implementing mitigation measures, however, will likely require time. To examine responses of the Southern Resident Killer Whale population to delayed implementation, I tested models with increases in the prey abundance starting either 5 years or 10 years from now. The following graph shows the mean projected Southern Resident Killer Whale population size when a 10% increase in Chinook is implemented immediately (top, blue line), after 5 years (middle, orange line), or after 10 years (bottom, red line). The long-term population growth rates after implementation again show that a 10% increase in prey is needed to stop the decline of Southern Resident Killer Whales. However, before that positive result is achieved, the population will have lost 4 whales if implementation takes 5 years, or 8 whales if implementation takes 10 years, relative to the expected population size if the increase in prey were achieved immediately. With positive growth of Southern Resident Killer Whale numbers after implementation of sufficient mitigation measures, a delay in implementation results in a loss of the potential initial years of recovery, and that lack of growth for those initial years leaves the population at a deficit in

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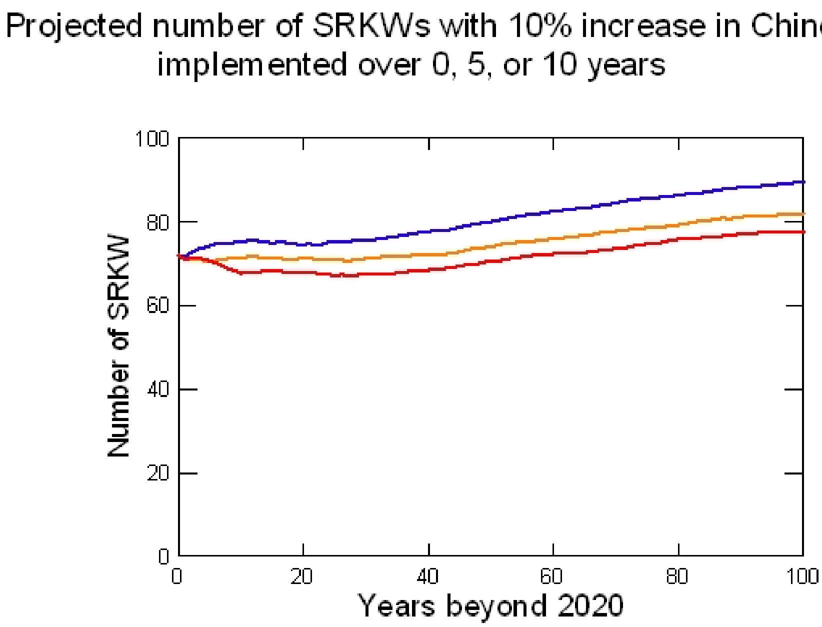
numbers throughout the subsequent recovery compared to what could have been. A 20% increase in Chinook allows for a long-term population growth of about 1% annually, but a delay of 5 or 10 years results in a loss of 8 or 16 whales before the growth begins, respectively, relative to the expected population size if growth had started in 2020.



32. In summary, although the 2019 SEAK BiOp does not provide management targets for slowing, stopping, or reversing the decline of the Southern Resident Killer Whale population, and it does not give specific estimates of the benefits to the Southern Resident Killer Whales of the proposed mitigation measures, for the above analyses I extracted from the 2019 SEAK BiOp what I could regarding the expected benefits of proposed actions. The 2019 SEAK BiOp provides various estimates of changes to Chinook stocks that might be expected from two of the mitigation measures – a reduction in the Southeast Alaska Chinook fishery as specified in the 2019 Pacific Salmon Treaty, and a proposed hatchery expansion – and it mentions other possible actions, such as habitat improvements, for which there is no quantification of expected results. Only if the additional, as yet unquantified, mitigation measures can boost Chinook abundance by

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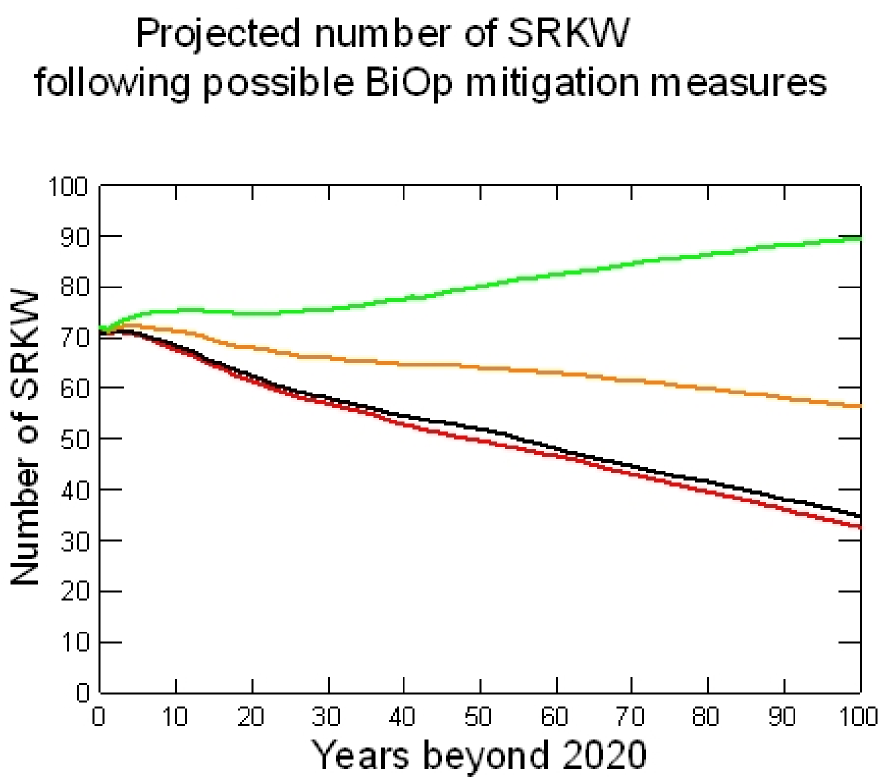
another 5%, would the combined effect of the proposed actions yield the 10% increase in Chinook that is necessary to halt the decline of the Southern Resident Killer Whales. The following graph summarizes the expected trajectory of the Southern Resident Killer Whale population if no changes are made from current conditions (bottom, red line), if a 0.5% increase in overall Chinook available to Southern Resident Killer Whales is produced by the reduced Chinook harvest in the 2019 Pacific Salmon Treaty (black line), if a 5% increase in Chinook is achieved by the hatchery mitigation (orange line), or if sufficient actions can be taken to achieve a 10% increase in Chinook (top, green line).



Conclusions

33. Based on previously published analyses, the results of updated models, my professional experience, and the information contained in the 2019 SEAK BiOp, I make the following conclusions with a reasonable degree of certainty:

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a. The Southern Resident Killer Whale population is in decline, and the projected status has deteriorated in just the past few years. The PVA models, using the latest available data on the current numbers, reproduction, and survival, project accurately the recent population changes.

b. The abundance of Chinook salmon prey available to the Southern Resident Killer Whales is a critical determinant of Southern Resident Killer Whale reproductive success and survival.

c. The mean Chinook abundance over recent years is not enough to allow reproduction by the Southern Resident Killer Whales sufficient to offset mortalities. An increase of about 10% in Chinook abundance would be required to stop the decline of Southern Resident Killer Whales, and an increase of about 35% in Chinook abundance would be required to achieve the healthy population growth rate of 2.3% that is the stated goal in the Southern Resident Killer Whale Recovery Plan.

d. The proposed mitigation measures in the 2019 SEAK BiOp have not been shown to be adequate to protect the future of the Southern Resident Killer Whale population – a short-coming that is admitted even within the 2019 SEAK BiOp. The quantitative estimates made in the 2019 SEAK BiOp would account for, at best and after full implementation, a reduction of half in the rate of decline in numbers of Southern Resident Killer Whales.

e. Full closure of the Southeast Alaska Chinook fishery, especially if combined with other mitigation measures, could result in enough prey to sustain a growing population of Southern Resident Killer Whales. Further enhancement measures

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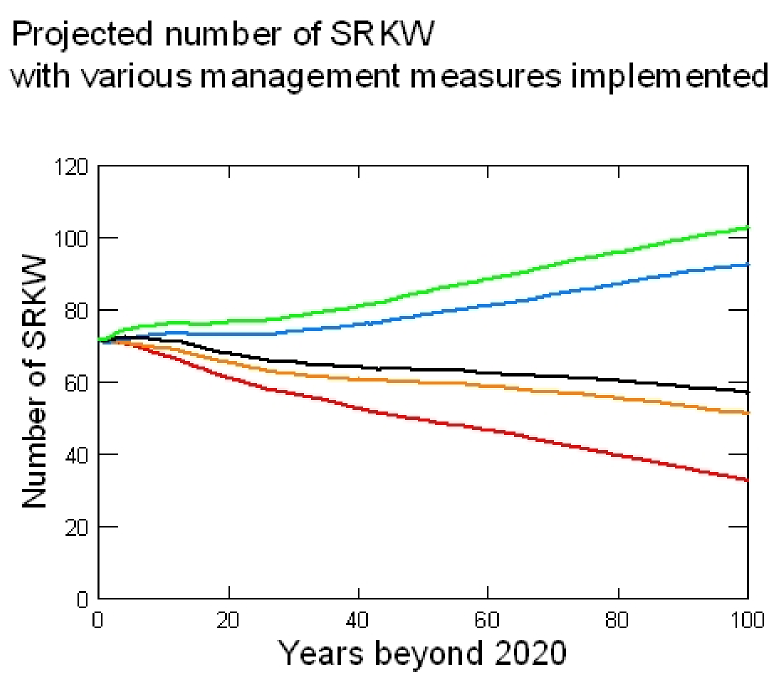
would be required to achieve the recovery goals set in the Recovery Plan for the Southern Resident Killer Whale. The last graph, below, shows projected Southern Resident Killer Whale numbers under current environmental conditions and management (bottom, red line), with the 5% increase in Chinook prey after 5 years, projected to result from the proposed hatchery enhancements (orange line), with a 6% increase in Chinook prey as might be achieved if the Southeast Alaska Chinook fishery is immediately closed (black line), with both the proposed hatchery project plus an additional 6% increase in Chinook abundance (blue line), or if a 12% increase in prey is achieved by the closure of the Southeast Alaska Chinook fishery (top, green line). The amount of increase in Chinook abundance as a result of reductions or closure of fishery harvests and other measures is uncertain, so responses of both the Chinook abundance and then the Southern Resident Killer Whale demography should be monitored closely, with adaptive management adjusting mitigation and enhancement measures as needed.



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I declare under penalty of perjury under the laws of the United States of America that the foregoing is true and accurate.

Executed this 15th day of April, 2020.

Robert C Lacy

Robert Lacy, Ph.D.

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UNITED STATES DISTRICT COURT
WESTERN DISTRICT OF WASHINGTON
AT SEATTLE

WILD FISH CONSERVANCY, Plaintiff, v. BARRY THOM, in his official capacity as Regional Administrator for the National Marine Fisheries Service, *et al.*, Defendants, and ALASKA TROLLERS ASSOCIATION, and STATE OF ALASKA, Defendant-Intervenors.	Case No. 2:20-cv-00417-RAJ-MLP **SECOND DECLARATION OF DR. ROBERT LACY, Ph.D.**

I, Robert Lacy, state and declare as follows;

1. I am over eighteen years of age. I have personal knowledge of the facts contained in this declaration and am otherwise competent to testify to the matters in this declaration.

2. I previously prepared a declaration that was submitted in this matter on April 16, 2020—Declaration of Dr. Robert Lacy, Ph.D, Dkt. No. 14-3 ("First Lacy Declaration"). The First Lacy Declaration described my professional qualifications and the work that I had

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performed and opinions that I had developed in this matter up to that point. I do not repeat those efforts here, but instead incorporation them with this reference.

3. In preparing this Second Lacy Declaration, I have considered the following additional materials not addressed in the First Lacy Declaration:

a. 2020 demographic data provided by the Center for Whale Research on births, deaths, and the current age structure of the Southern Resident Killer Whale population;

b. A report of new analyses by Fisheries & Oceans Canada on the impacts of Chinook abundance and other threats on Southern and Northern Resident Killer Whales that has been accepted by the scientific journal Biological Conservation and will be available as an on-line publication within the next week or two (Murray, C.C., et al. 2021. "A cumulative effects model for population trajectories of resident killer whales in the Northeast Pacific" Biological Conservation);

c. Published analysis from a research team led by National Marine Fisheries Services ("NMFS") scientists of the species and stock composition of the prey used by the Southern Resident Killer Whales (Hanson, M.B., et al. 2021. "Endangered predators and endangered prey: Seasonal diet of Southern Resident killer whale" PlosOne 16:e0247031).

4. I have conducted further analyses using these subsequently developed data. This declaration is intended to supplement the opinions expressed in the First Lacy Declaration to describe those new efforts. Except as expressly stated herein, I continue to hold the opinions described in the First Lacy Declaration.

5. This declaration also responds to various remarks on the First Lacy Declaration contained in the Declaration of Lynne Barre, Dkt. No. 43-3 ("Barre Declaration").

6. In summary, the opinions I express herein are as follows:

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a. Analyses that make use of the most recent data on the Southern Resident Killer Whale population reinforce my earlier conclusions that the population is projected to be in slow decline and that increases in Chinook prey availability could stop the decline and allow the population to recover.

b. Using the most recent killer whale demographic data, I now estimate that a 5% increase in Chinook abundance, which is the maximum increase that NMFS claims could result if the proposed increases in hatchery production were fully implemented, would be sufficient to stop the decline, but would not be sufficient to support the growing population called for in the Recovery Plan. Therefore, reductions or modifications of Chinook harvest would be necessary to provide the level of increase in abundance of the preferred prey of the Southern Resident Killer Whales that is needed to allow growth and recovery of the killer whale population.

c. If recent estimates of weaker relationships of Resident Killer Whale birth and death rates on Chinook abundance are incorporated into the population projections, I would then estimate that the Chinook available to the killer whales would need to increase by more than 10% to stop the decline of the Southern Resident Killer Whales.

d. The Barre Declaration incorrectly states that the analyses of effects of changing abundance of Chinook depend on an assumption that all Chinook that escape from the fishery would be consumed by Southern Resident Killer Whales.

e. The Barre Declaration incorrectly asserts that the analyses presented in the First Lacy Declaration include the effect of Chinook prey as the only factor influencing the Southern Resident Killer Whale population growth and recovery.

f. The Barre Declaration criticizes the population analyses in the First Lacy Declaration for using outdated estimates of the correlation between prey abundance and killer

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whale birth and death rates and asserts that newer analyses show weaker effects. The Barre Declaration did not identify any such revised analyses available in peer-reviewed scientific publications as of the time of the First Lacy Declaration. However, as this Second Lacy Declaration was being prepared, a report based on recent analyses of the Fisheries and Oceans Canada (to which I was a contributor) was formally accepted by the journal Biological Conservation and will be published this month. That report found relationships of Southern Resident Killer Whale birth and death rates to Chinook abundance that are weaker than had been reported previously, but the report notes again that Chinook abundance has the largest effect of those factors that have been identified as possible threats to the Southern Resident Killer Whale population. As noted in paragraph 6.c, above, if the relationship of killer whale demography to Chinook abundance is weaker than previously estimated, then actions that result in greater increases in Chinook available to Southern Resident Killer Whales will be required to achieve population recovery.

Updated Population Viability Analyses

7. The analyses presented in First Lacy Declaration were based on the long-term, detailed records of births and deaths in the Southern Resident Killer Whale population from 1976 through December 2019. Those years of monitoring include periods of population growth (e.g., 1984-1994) as well as periods of decline (e.g., 1995-2001, and most recently 2016-2019). Projections were made of population growth expected for the population as it existed at the beginning of 2020, under a variety of scenarios of possibly improved levels of the Chinook prey abundance. Fortunately, since those analyses were completed, the population has increased by one in 2020, due to two births (a male and a female that are still living) and one death (a 43-year-old male). Another birth has occurred in early 2021.

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8. I have added those recent demographic events to the data and re-calculated the long-term mean birth and death rates for the years 1976 through the end of 2020. (The 2021 birth was not included in these calculations of birth and death rates because it is not yet known if it will survive into the next year, and we do not yet have data on the full 2021 year of reproduction.) I then repeated the same population projections as were presented in First Lacy Declaration, but now with the updated birth and death rates and projecting forward from the population as it exists as of March 2021. With these updated analyses, I now project an average rate of decline of 0.4%. This is slightly slower than the 1% annual decline projected a year earlier in the First Lacy Declaration. I now estimate that the probability of the population becoming functionally extinct with fewer than 30 animals during the next 100 years is 21%, compared to the 59% estimated a year ago. These changes result from the available data now including the better calf survival of the past year, no deaths of females in 2020, and a slightly larger current population. The current projections fall between prior estimates made from data through 2015 and the estimate made from data through 2019, as expected since the population declined in the years 2016-2019, but slightly recovered in 2020. I caution, however, that fluctuations in births and deaths from year to year are expected, and short-term changes in the population should not be assumed to be indicative of long-term trends. An advantage of the population viability analysis models that I and others use is that the models can indicate the long-term consequences of historic patterns and known or projected changes to the habitat. A common definition of endangerment used in Endangered Species designations by the U.S. Fish & Wildlife Service is that there is a greater than 5% probability of extinction within 100 years. The current risk to the Southern Resident Killer Whale far exceeds this threshold, even with the small improvement observed in 2020.

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9. I repeated the analyses of population projections with several levels of increased Chinook salmon abundance. With the newest demographic data included, I now estimate that prey availability would need to increase about 5% relative to the long-term (1976-2020) average in order to stop the long-term decline of the Southern Resident Killer Whale (i.e., to achieve zero population growth), and Chinook would need to increase about 30% to result in the 2.3% growth specified for delisting in the species' Recovery Plan. These estimates are marginally more optimistic than the estimated 10% and 35% more Chinook that were calculated a year ago for halting the decline and achieving recovery, respectively.

10. The 2019 biological opinion on Southeast Alaska salmon fisheries ("2019 SEAK BiOp") discussed possible mitigation measures that would attempt to increase the prey availability for Southern Resident Killer Whales. The 2019 SEAK BiOp estimates the newly negotiated 2019 Pacific Salmon Treaty will reduce the Southeast Alaska fishery annual harvest of Chinook by up to 7.5% relative to the harvest under the 2009 Treaty. A proposed increase in hatchery production mitigation seeks to provide 4% to 5% increase in prey available to the Southern Resident Killer Whales. The increase in hatchery production is not yet funded, so I would expect a delay of at least 5 to 10 years to account for allocation of funds, construction of any new facilities, increased programs of production, and then growth of hatchery-raised Chinook to the size preferred by killer whales as prey.

11. I applied these estimates from the 2019 SEAK BiOp to the population viability analysis ("PVA") model, with the updated estimates of demographic rates. The estimated 7.5% (maximum) reduction in the Southeast Alaska fishery, applied to a typical 6% reduction in prey available to the Southern Resident Killer Whales caused by the Southeast Alaska fishery as a whole (the 6% being an approximate middle value from the many estimates made in the SEAK

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BiOp), results in a less than 0.5% increase in the Southern Resident Killer Whale prey. This is only 1/10th of the 5% increase that is now projected to be needed to achieve even a cessation of the decline in Southern Resident Killer Whale population.

12. To estimate the possible reductions in threats to the Southern Resident Killer Whale that might be achieved with greater reductions in Chinook fisheries, I used the updated estimates to project the Southern Resident Killer Whale population growth with an immediate 6% increase in Chinook prey, and a 3% and a 12% increase in prey (half and double the middle estimate, covering most of the range of values reported in the 2019 SEAK BiOp for specific stocks and years). As shown in the following graph, with the existing baseline in red (bottom line), the PVA projections for these scenarios show that the 3% increase in Chinook results in a mean 0.1% decline in Southern Resident Killer Whale population per year (blue line), the 6% increase in Chinook results in a slow 0.2% increase of the Southern Resident Killer Whale population (green line), and the 12% increase results in 0.7% positive growth annually (top, black line). Thus, adequate prey to support growth of the Southern Resident Killer Whale population could be achieved by reductions in harvest of Chinook, whether from the SEAK fishery or other fisheries that impact Chinook stocks utilized by the killer whales.

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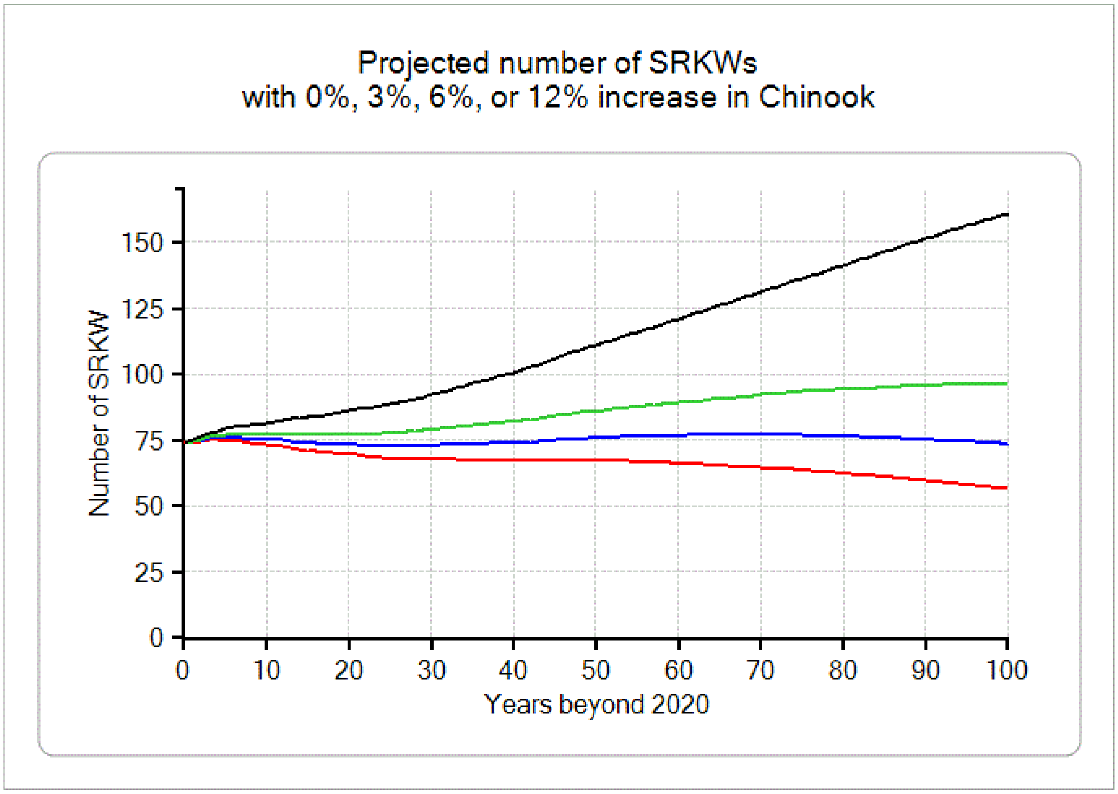

13. With the updated demographic calculations, if the proposed hatchery expansion identified by NMFS in the 2019 SEAK BiOp were fully implemented and achieved the maximum increase in prey predicted by NMFS of 5%, Chinook available to the Southern Resident Killer Whale might be just sufficient to halt the decline, but still not allow recovery, of the Southern Resident Killer Whale. NMFS predicts a delay of 5 to 10 years in achieving the increase in prey through hatchery production due to time required to implement programs and for the released Chinook to mature. The Southern Resident Killer Whale population is predicted to lose about 2 or 4 whales over the 5 or 10 years, respectively, during that period.

Responses to Statements Made in the Barre Declaration

14. The Barre Declaration contains statements about the modeling and conclusions in the First Lacy Declaration that are misleading, inaccurate, or actually under-cut NMFS's position

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that refined analyses would not support my conclusions.

15. It is stated in several places in the Barre Declaration that the changes in the abundance of Chinook salmon available to Southern Resident Killer Whales would have a smaller effect than projected because "the SRKW would not intercept and consume all of those fish," "the fish are subject to other predators and sources of mortality," and "not all the fish escaping the fishery and migrating south would be intercepted by or consumed by the whales." However, no one claims that all the fish escaping the fishery would be consumed by the whales, and it is illogical to assert that such an assumption is necessary in order to estimate the impacts on killer whales of a change in overall abundance of the primary prey. A given percent change in the Chinook available to Southern Resident Killer Whales (for example, a 5% increase) will result in the same percent change in the Chinook eaten by the Southern Resident Killer Whales, regardless of whether the killer whales consume 1%, 10%, 25%, or any particular proportion of the total prey abundance, unless other factors (such as the efficiency with which killer whales can catch salmon) also change. Moreover, the demographic calculations on which the population projections are based estimate the relationships between the demographic rates experienced by the Southern Resident Killer Whales and indices of the overall Chinook abundance, not relationships to the unknown number of Chinook salmon actually consumed by Southern Resident Killer Whales. Importantly, the multiple analyses that have shown impacts of Chinook abundance on Southern Resident Killer Whale birth and death rates – including the most recently published analyses – demonstrate that the killer whales do not adjust their feeding behavior in a way that fully offsets the effect of changes in Chinook abundance.

16. The Barre Declaration incorrectly states that my analyses focus "on SEAK fisheries alone as the only factor influencing recovery of the Southern Resident Killer Whale

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population." As acknowledged in the 2019 SEAK BiOp, my population projections specifically include impacts of boat disturbance and PCBs, the two other factors identified by NMFS and others as primary threats to the Southern Resident Killer Whale. For example, with the structure of the model and the parameter estimates that I used, a delay or slowing of reproduction by females will be predicted to lead to greater accumulation of PCBs in their tissues and therefore reduced survival of their calves. Conversely, increased reproduction will lead to reduced PCB loads in adult females (due to depuration via lactation) and therefore improved calf survival. A powerful advantage to the Population Viability Analysis modeling approach that I and others have used is that it can incorporate such cumulative and interacting impacts of multiple threats and allow statistical analysis of the relative role of each threat in influencing population growth and recovery. My multiple papers and reports on the factors influencing Southern Resident Killer Whales and the benefits of various possible management actions all take this approach and conclude that the impact of Chinook abundance is greater than that of the other identified threats. The First Lacy Declaration focuses on the Chinook availability because that is the factor being addressed in the 2019 SEAK BiOp and in this case. I have at other times and for other agencies presented findings from my analyses related to impacts of other factors such as PCBs, boat disturbance, and oil spills when those factors were being addressed in resource management and recovery plans, and I have shown that a strategy that combines improvements to Chinook abundance with reductions in noise and PCBs can achieve faster recovery than would a focus on Chinook prey abundance alone (Lacy et al. 2017).

17. The Barre Declaration states "their conclusion that prey is most important is highly dependent on the assumptions and inputs to the model and their reliance on outdated correlations between prey abundance and whale vital rates." However, all models and all

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analyses, including those of NMFS, are necessarily dependent on the assumptions and inputs. What is relevant is whether the analyses use the best, documented sources of data to provide those inputs. At the time of the First Lacy Declaration and the Barre Declaration, I used the estimated relationships between prey abundance and Southern Resident Killer Whale demographic rates that had been published in peer-reviewed scientific literature. A report of Fisheries and Oceans Canada had presented new analyses of a working group, in which I participated (Murray et al. 2019. Cumulative Effects Assessment for Northern and Southern Resident Killer Whale Populations in the Northeast Pacific. DFO Can. Sci. Advis. Sec. Res. Doc. 2019/056. x. + 88 p). That report found weaker relationships of Chinook abundance to Resident Killer Whale birth and deaths rates in recent years, but again concludes that "prey availability was the most important threat for these populations [the Northern and Southern Resident Killer Whales] followed by vessel noise/presence" and that the findings "strongly support the significant role of prey availability in determining the population trajectory of these populations, and are consistent with previous work." However, that document acknowledges in its Foreword that it is "not intended as definitive statements on the subjects addressed but rather as progress reports on ongoing investigations." Therefore, I did not incorporate those provisional estimates of impacts of prey availability into my analyses for the First Lacy Declaration.

18. The findings in the report of Fisheries and Oceans Canada (after some corrections to the tabular display of calculations) have recently been accepted (on 2 April 2021) and will be published this month (Murray et al. 2021. Biological Conservation). These recent estimates of impacts of prey availability would have no effect on the projections for the Southern Resident Killer Whale population under current conditions, because that projection of slow decline is based on the long-term average birth and death rates and an assumption that the availability of

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Chinook to the killer whales will remain, on average, as it has been in recent decades (1976-2020). However, if we use the weaker relationships reported in Murray et al. (2021), rather than the relationships estimated in other scientific studies on which I relied previously, then projections of Southern Resident Killer Whale population growth under improved stocks of Chinook will necessarily show lesser benefits would be achieved by any given percent increase in Chinook. If I use the estimates of reduced prey effects, I calculate that at least a 10% improvement in the mean abundance of Chinook available to the Southern Resident Killer Whales would be necessary to stop the long-term decline in the killer whale population. This is about the same as was estimated in the First Lacy Declaration, and it again indicates that the management actions proposed by NMFS are not projected to be adequate to stop the decline in the population. With these revised model inputs, I calculate that a 12% improvement in Chinook abundance would be required to reduce the probability of extinction from the currently estimated 21% to the 5% that would indicate escape from endangerment.

19. The Barre Declaration notes that "correlation does not mean causation" and suggests therefore that changing Chinook abundance might not affect the Southern Resident Killer Whales. This assertion is counter to prior statements by NMFS, and it overlooks that multiple studies using varied methodologies and data sets, including the most recent analysis by Fisheries and Oceans Canada, have found both that Chinook abundance influences Southern Resident Killer Whale demography and that the Chinook abundance has the largest effect of those factors that have been identified as possible threats to the population. Moreover, virtually all species recovery plans are based on the logical conclusion that when other documented possible causes of responses have been removed through statistical analysis, then observed correlations are our best indication that a causal relationship exists and should be the focus of

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management action. Otherwise, no actions to protect and recover the Southern Resident Killer Whale (or any endangered species) would ever be taken, because it is not possible to do the experimental manipulations (with adequate sample size, replication, and controls) that would be necessary to definitively prove causation.

20. The Barre Declaration also asserts that "more recent data shows that the correlations have weakened." However, those findings were not yet available in peer-reviewed scientific literature at the time of the First Lacy Declaration, the Barre Declaration did not present any such revised statistical analyses, and each study of which I am aware comes to the conclusion that the prey availability is the factor with the largest impact on the Southern Resident Killer Whale. As noted above (paragraph 18), using the results from the pending publication of a study that show weaker (but not zero) correlations does not change the overall conclusion that improving Chinook abundance can achieve recovery of the Southern Resident Killer Whale population, but only if the Chinook abundance is increased by more than NMFS estimates will be achieved by the actions described in the 2019 SEAK BiOp.

21. Moreover, in March 2021 NMFS scientists and other scientists published an extensive analysis of the seasonal diet of Southern Resident Killer Whales (Hanson et al. 2021). Regarding the statistical association between prey abundance and killer whale fecundity, Hanson and colleagues cite the same documents (Ward et al. 2009 and Ford et al. 2010) that provided the correlations between Chinook abundance and killer whale demography that Barre dismisses as "outdated." Hanson et al. go on to state "… our finding that Chinook salmon prominently appeared in the diet year-round suggests this relationship may be causal." Dr. Barre was a co-author of this paper, and it neither provides nor relies upon any more recent data.

22. Published analyses of the correlations between prey abundance and killer whale

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demography all note that it is difficult to know which salmon stocks are most important to the Southern Resident Killer Whale. Those studies concur, however, that aggregate indices of multiple stocks of Chinook provide the best predictor of Southern Resident Killer Whale demographic rates. To the extent that these statistical analyses failed to identify the specific mix of Chinook stocks that are most important to the Southern Resident Killer Whales, the analyses would under-estimate the strength of the true relationships. Importantly, if the correlations have recently become weaker, revised analyses show that greater increases in prey abundance will be needed in order to stop the decline and achieve recovery. The arguments made in the Barre Declaration for a weaker correlation would suggest that the projections I made are overly optimistic and that prey availability would need to increase more that I estimated in order for the Southern Resident Killer Whale population to stabilize or achieve recovery.

23. The Barre Declaration asserts that the better forecasted salmon abundance in 2020 shows that the Southeast Alaska exclusive economic zone "fishery would not have any meaningful effect on the health or status of any individual SRKW" and states "[i]n the 2019 biological opinion NMFS concluded the SEAK fisheries would not appreciably reduce reproduction or survival and would not jeopardize the SRKW. This finding remains valid in light of our recent analysis for 2020." Given the acknowledged high variability and unpredictability of salmon abundance in any given year, the expectation for a single year cannot provide strong support for any conclusions. However, the improved demography of the Southern Resident Killer Whale in 2020, compared to the prior four years, coincided with that forecasted better abundance of prey, which would support the contention that the Southern Resident Killer Whales are significantly affected by prey abundance.

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I declare under penalty of perjury under the laws of the United States of America that the foregoing is true and accurate.

Executed this 3rd day of May, 2021.

Robert C Lacy

____

Robert Lacy, Ph.D.

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