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Rebuilding the Georgia Strait sport fishery through marine mammal culling

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Abstract

The Georgia Strait sport fishery has declined dramatically over the last 30 years. There is strong correlative evidence that this decline has been due to impact of growing harbour seal abundance on the first-year marine survival rates of Chinook and Coho, before the fish reach sizes attractive to anglers. Fishing effort has responded strongly to the decline in abundance of larger fish, to result in a nearly linear trade off relationship between effort and seal abundance. Simple models based on the historical data predict that the fishery could be largely restored through moderate (50%) reductions in the seal population, possibly by allowing First Nations harvesting as a traditional use and right.

Introduction

Historically, the Georgia Strait supported one of Canada's largest marine recreational fisheries, mainly for Chinook and Coho salmon. At its peak during the 1970s, the fishery had close to 800,000 boat days per year, providing substantial benefits to communities around the Strait in terms of local recreational opportunity and economic activity related to tourism. The fishery was initially based on a large number of wild stocks from around the Strait and Puget Sound, supplemented by rapid growth in hatchery production during the 1970s. Roughly 50% of the Chinook catch has been hatchery fish since hatchery production peaked in the early 1980s (Beamish et al. 1995). The percentage contribution of hatchery Coho has increased dramatically, from near 30% in the early 1980s to over 70% for some years since 2000 (Sweeting et al. 2003; Beamish et al. 2008).

But signs of decline began to appear in the late 1970s (Fig. 1), particularly of Chinook stocks, and an analysis of management options by Argue et al. (1983) warned of possible overfishing due to high fishing efforts attracted by abundance of hatchery fish. The decline accelerated through the 1980s, prompting closure of the commercial troll fishery and various regulations (size and bag limits, local closures) of the sport fishery. None of these regulatory actions had any apparent effect on the decline, and the fishery had severely declined by the late 1990s when there was complete closure to retention of wild Coho salmon (Neville et al. 2015).
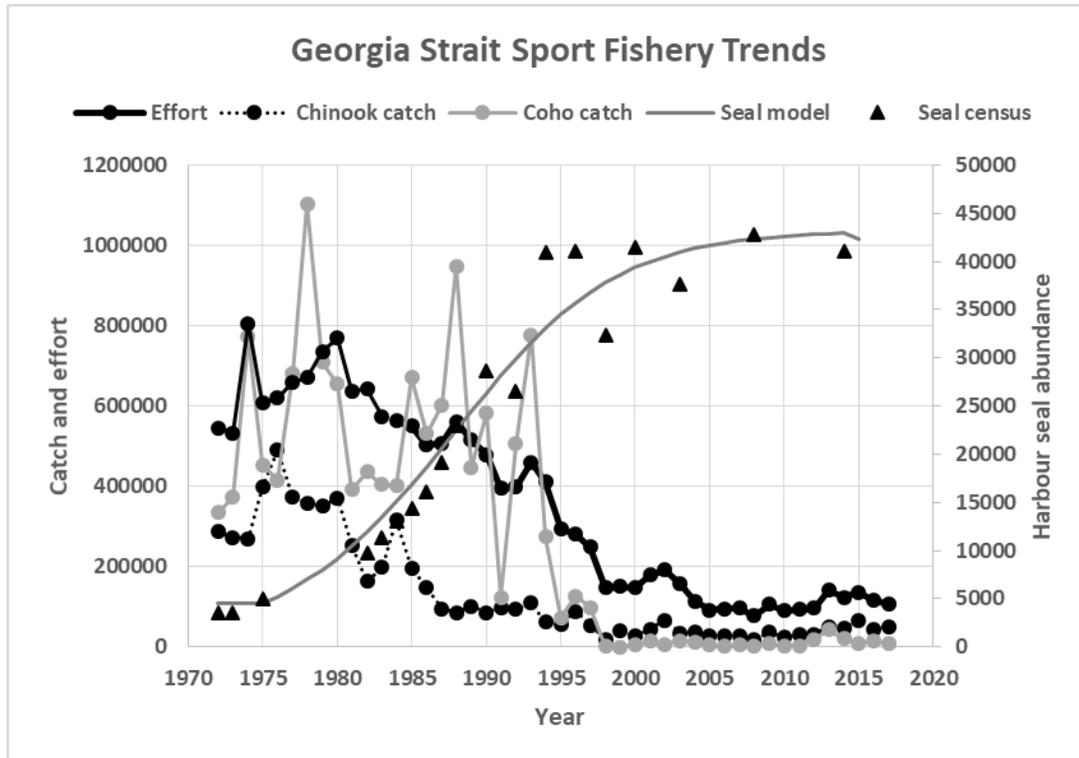
Figure 1. Trends in sport fishing effort and catches in the Georgia Strait sport fishery, and trends in harbour seal abundance reported by Olesiuk (2010). Smooth curve for seals is fitted logistic growth model with r=0.17, K=43000.

By the early 1990s, trend data on first ocean year survival rates of juvenile Chinook and Coho had become available from intensive coded wire tagging (CWT) programs (aimed initially at evaluation of hatchery performance, but also for a few wild indicator stocks), and it was obvious from these data that there had been a severe and persistent decline in the proportion of smolts surviving to reach harvestable sizes (Fig. 2, and see recent summaries by Zimmerman et al. 2015 and Ruff et al. 2017). The number of smolts entering the Strait has not declined substantially (as would be expected if there were an overfishing or freshwater habitat problem), for either indicator wild populations or hatchery releases.
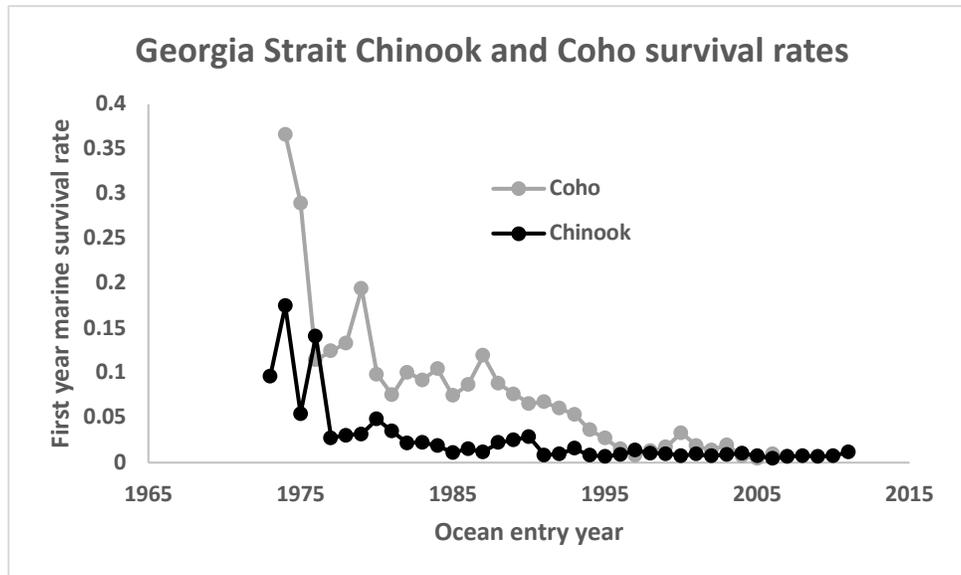
Figure 2. First ocean year marine survival rates for Chinook and Coho salmon in the Georgia Strait, estimated from coded wire tagging data. Numbers shown are averages over hatchery and wild indicator stocks. Estimates for the first few years are probably biased upward due to various sampling problems while the CWT recovery program was developing.

Over the years, a variety of oceanographic and trophic factors have been used to explain the declines in early ocean survival of Chinook and Coho, including changes in water temperature, river flow, wind patterns, seal abundance, and herring abundance (Beamish et al. 1995, 2008, 2010; Tanasichuck et al. 2008). But none of those correlative studies (except correlations with seal abundance found by Tanasichuck et al.) has convincingly stood the test of time, and there have been no clear "bottom up" trends in food availability as indicated by plankton data (Mackas et al 2013). One multivariate correlative study shows an apparent bias toward "bottom up" explanations based on oceanographic factors; Araujo et al. (2013) claim to have examined seal abundance as a survival covariate, but do not include that factor in their final multivariate model structure. Two ecosystem modeling exercises (Martell et al. 2001,Preikshot et al. 2013) have tried to predict combined impacts of fishing, ocean changes, and marine mammal abundances. To complicate matters, Coho catches have varied more violently than predicted from changes in overall marine survival rate. This variation has apparently been due to changes in the proportion of Coho rearing in the Georgia Strait. Prior to the early 1990s, coded wire tagging data showed a high proportion of the catch from Georgia Strait Coho stocks to occur within the Strait. But beginning in the early 1990s, Beamish et al. (1999) document increasing variability in that rearing proportion and an increase in the proportion of fish rearing outside the strait; they attributed the change dispersal/migration patterns to climatic factors (Fraser River flow, wind patterns).

We propose that the main cause of the first year marine survival decline, hence of the fishery, has been the explosive growth of harbour seal populations that followed marine mammal protection in the region (Fig. 1). We are certainly not the first to propose large marine mammal impacts on salmon in the region, see e.g. Chasco et al. (2017a,b). Lessard et al. (2005), and Tanasichuck et al. (2008). In the following sections, we first review correlative and functional evidence that seals have caused large increases in first year mortality rates. Then we propose a simple model for predicting how survival rates and fishing efforts would respond to managed reductions (harvest or culling) of seals. We conclude that moderate reductions in the seal population would result in a

major recovery of the fishery, and other conservation and harvest management benefits from stocks that have severely declined and now limit harvest opportunities even outside the Georgia Strait (in particular on the West Coast of Vancouver Island).

Estimation of marine mammal impact on early ocean survival

Four lines of evidence support the argument that marine mammal predation has been the main cause of marine survival declines. The first two are correlative, from marine survival rate and stock-recruitment data. The third is from expansion of seal diet composition estimates using bioenergetics models. The fourth is from analysis of possible encounter rates between seals and juvenile salmon based on searching behavior of seals.

The simplest line of evidence for major impact of seals comes from simple correlation analysis. When predicting survival changes, it is often best to do so by predicting not survival rate directly, but rather the instantaneous natural mortality rate M (Bradford 1995), which is just $-\ln$(survival rate). Fig. 3 shows the historical relationship between first ocean year M (calculated from survival rates in Fig. 2) and harbour seal abundance (Fig. 1). The relationship is obviously nearly linear and very strong, indicating constant incremental impact per seal on the overall instantaneous mortality risk. Such a linear relationship is expected if seals do not target juvenile Chinook and Coho (and the juveniles are not a high proportion of the seal diet), but instead take them when "randomly" encountered while searching for other prey. Seals do target Chinook and Coho smolts in some places when these smolts are concentrated during estuarine migration, but that targeted feeding likely takes only a small total percentage of the smolts.
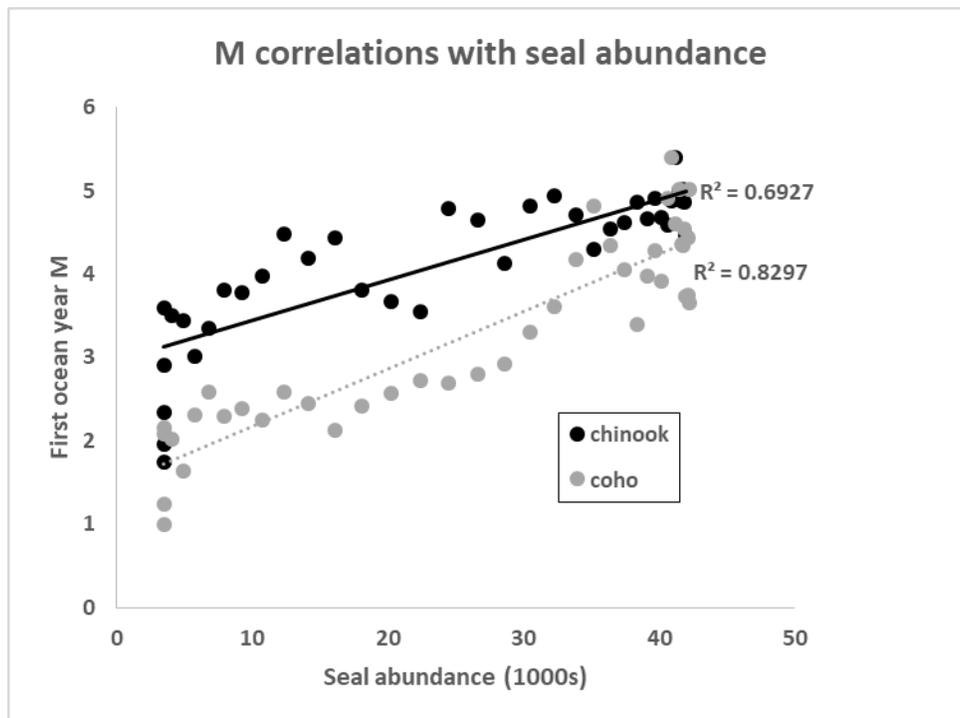

Figure 3. Correlative relationship between first year instantaneous natural mortality rate (M) estimated from coded wire tag data (as $-\ln$(survival rate)), and abundance of harbour seals in the Georgia Strait. Each data point represents one year, using average survival rates from Fig. 2 and logistic population model estimates of seal abundance from Fig. 1.

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A second correlative line of evidence is from analysis of Chinook stock-recruitment data (Nelson et al in prep). Plots of log recruits per spawner (lnR/S) against spawner abundance for a collection of Georgia Strait and Puget Sound stocks show that seal abundance explains a high proportion of the observed decline over time in lnR/S, implying about the same increase in total pre-recruit M as we would predict from the CWT data in Fig. 3 (an increase in M of about 2.0 over the range of observed seal abundances).

The third line of evidence is from seal diet studies in 2012-3 (Thomas et al. 2016) and expansion of diet estimates to total juvenile consumption using bioenergetics models (Nelson, ms in prep.), similar to the approach taken by Chasco et al. (2017). Based on the diet data and seal food consumption rates estimated by Olesiuk (1993), we calculate that seals have recently been consuming around 3-5 million juvenile Coho from the annual smolt run of about 10 million, and around 15-20 million juvenile Chinook from the annual smolt run of around 50 million, both mainly over the period May to October. To convert these consumptions to components of the total first year mortality rate M, we need to divide the year into components before, during, and after the main period of predation (April-September), while accounting for both estimated seal consumption and size-dependent natural mortality due to factors other than seals from the number of juveniles at risk to predation. Using size-dependent natural mortality rates inversely proportional to fish length as recommended by Lorenzen (1996) and adjusted so as to give observed average survival rates from the late 1970s when seals were still not abundant, we estimate that the increment in first year M for both Coho and Chinook due to the estimated seal consumptions above has been in the range 1.3-3.0, with the most likely incremental change being near the value of 2.0 implied by the changes in M shown in Figure 2. There is high uncertainty about the inputs to this calculation (number of smolts, number eaten, mortality rates before and after the predation period), leading to wide posterior distributions for the seal predation component of total M (Figure 4).
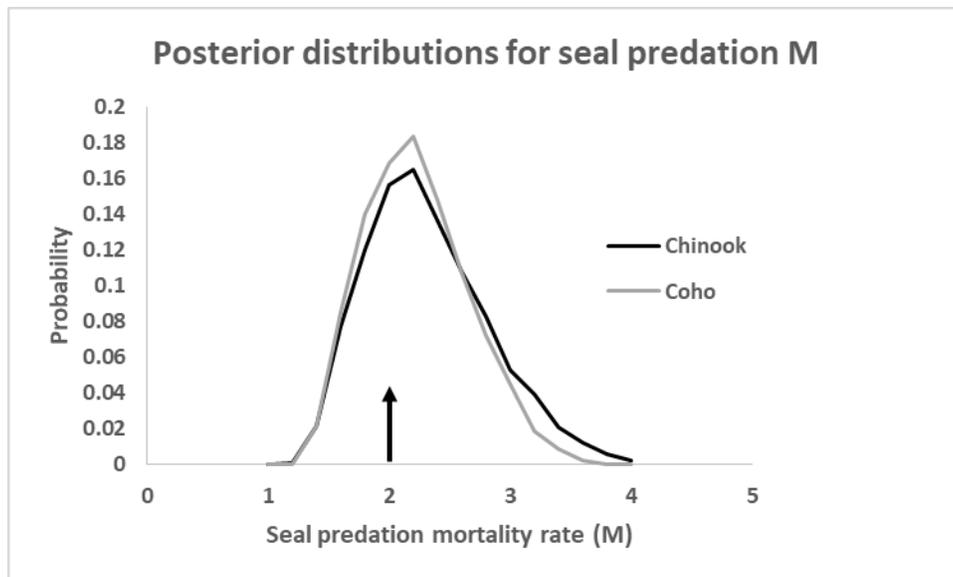

Figure 4. Posterior distributions for seal predation rate in recent years (2012-3) based on estimates of total smolts consumed from seal diet data.

The fourth line of evidence is from analysis of seal foraging behavior, to provide at least an upper bound on the daily risk of juvenile salmon being encountered by seals, as the volume of water searched per day by foraging seals divided by the volume of water over which the juvenile salmon

are distributed.   That calculation requires estimates of seal foraging speeds, reaction distances to prey, and proportion of time spent feeding in the part of the water column (mainly top 30m; C. Neville, DFO, pers. comm.) where the juveniles are mainly found. An example calculation (Table 1) using foraging speed and foraging time estimates from a recent study in the central Georgia Strait by Allegue (2017) suggests that the daily encounter risk for a randomly distributed juvenile salmon in the Strait may be as high as 0.01, implying a potential daily predation rate of 1% per day. This implies that the cumulative M over the June-September period when juveniles are most common in the diet (Thomas et al. 2016) could easily be at least as high as 120 x 0.01 = 1.2, somewhat lower than the cumulative increase in M of around 2.0 indicated by the data in Fig. 3 (but that M increase is estimated for the whole first year of ocean residence).

Table 1. Component assumptions and calculations in estimation of potential seal encounter (and mortality) risk per day for juvenile salmon swimming at random in the Georgia Strait

Component of calculation	Estimate	Source
(1) Seal swimming speed (m/sec)	1	seal tracking data, various sources
(2) Seal reactive distance (m)	5	guess based on sensory (visual, smell) acuity
(3) Potential volume searched/day (m³)	6785840	π x (1) x (2)^2 x seconds/day
(4) Volume of top 30m (m3) per km²	30000000	
(5) Potential swept volume/surface volume/seal	0.226	ratio of (3) to (4)
(6) Proportion of day spent surface foraging	0.025	From Allegue (2017) foraging time allocation
(7) Daily encounter risk per seal per km²	0.00565487	product of (5) and (6)
(8) Seal density (numbers/km²)	2	from census data and Georgia Strait area
(9) Daily encounter risk	0.011	product of (7) and (8)
(10) 120 day summer mortality risk (M)	1.36	120 times (9)

It must be emphasized that two of the key parameters used in the Table 1 calculation are highly uncertain: we know little about reactive distances of seals to smaller prey items like juvenile salmon, and the proportion of time that seals spend foraging in surface (top 30m) waters may be highly variable in time and space depending on factors ranging from haulout behaviors to availability of alternative prey in different habitat types. Allegue (2017) saw three main foraging modes: estuarine (targeting salmon), day-time surface, and night-time deep (at bottom), and at least the time allocation between surface and deep foraging modes may be highly unpredictable.

Response of fishing effort to declining fish abundance

There has apparently been a strong, logistic response of fishing effort to declines in Chinook and Coho harvestable abundance (Fig. 5). But exploitation rates have not decreased as rapidly as predicted from decreases in effort assuming constant catchability (q), indicating that the fishermen still pursuing Chinook and Coho are now much more efficient than the average anglers of the 1970s.
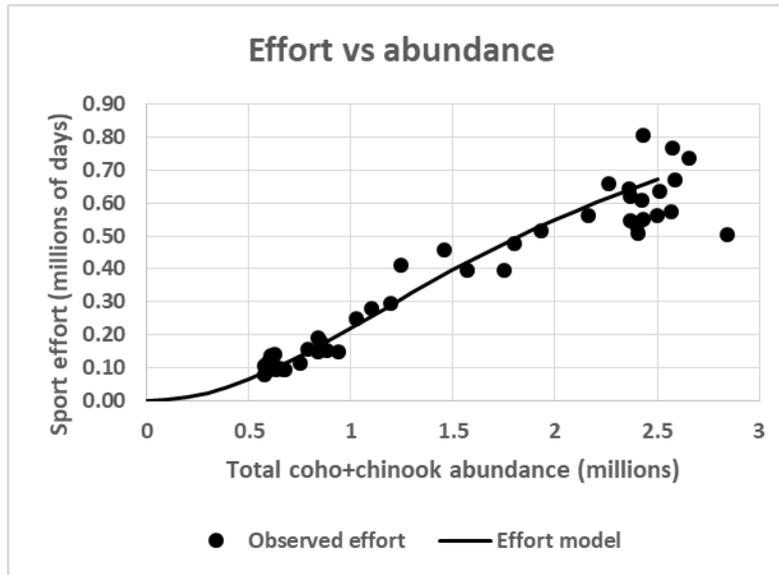
Figure 5. Historical relationship between sport fishing effort and estimated abundance of harvestable Chinook and Coho in the Georgia Strait. Logistic response relationship fitted by eye. Relative catchability change estimated so as to fit estimated changes in exploitation rate over time from CWT data. Abundance estimated for each year as average wild smolt production plus hatchery smolt production multiplied by first year marine survival rate estimated from CWT data; Chinook abundance corrected to include probable numbers of age 3 and 4 fish based on exploitation rates and annual survival rate of 0.7.

The beginnings of this effort response pattern, i.e. a weak positive relationship between effort and fish abundance, was first noticed by Argue et al. (1983). Such responses are generally expected in sport fisheries (Post et al. 2002), where progressively more fishermen choose other recreational activities as success rates decline due to reduced fish density. Note that catch per effort is typically not a good predictor of the effort response; cpue has not declined that much in the Georgia Strait (i.e. cpue has been "hyperstable" as noted by Peterman and Steer, 1981), most likely because less capable fishermen drop out of the fishery first so that the people who are still fishing today are those with higher individual catchability coefficients (experienced, better fishing techniques, e.g. fishing guides; see Van Poorten et al. 2016).

Model for prediction of culling impact on future fishing effort

Based on the information presented above, it appears that we can make reasonable predictions about the likely impact of seal culling policies on future sport effort in the Georgia Strait by using a simple model with just four basic assumptions:

(1) Chinook and Coho smolt inputs to the Georgia Strait from wild populations and hatcheries will continue to be high (i.e. there will not be recruitment overfishing of wild stocks, or major freshwater habitat deterioration).

(2) The seal population will continue to exhibit logistic population growth so as to "push back" against impacts of culling removals as the population has in the past (i.e. seals will not collapse naturally due to other factors like general depletion of their alternative prey or predation by transient killer whales).

(3) First year ocean mortality rate of juvenile Chinook and Coho will continue to be linearly related to seal abundance as in Figure 4.

(4) Sport fishing effort will continue to exhibit a logistic response to combined Chinook and Coho ocean age 2 and older abundance (i.e. there has not been an irreversible loss of interest and capability to pursue ocean sport fishing).

These assumptions are easily captured in a spreadsheet format that can be run forward over time beginning back around 1970, with historical changes in hatchery smolt production, to check retrospective fit to the data (Figure 6).
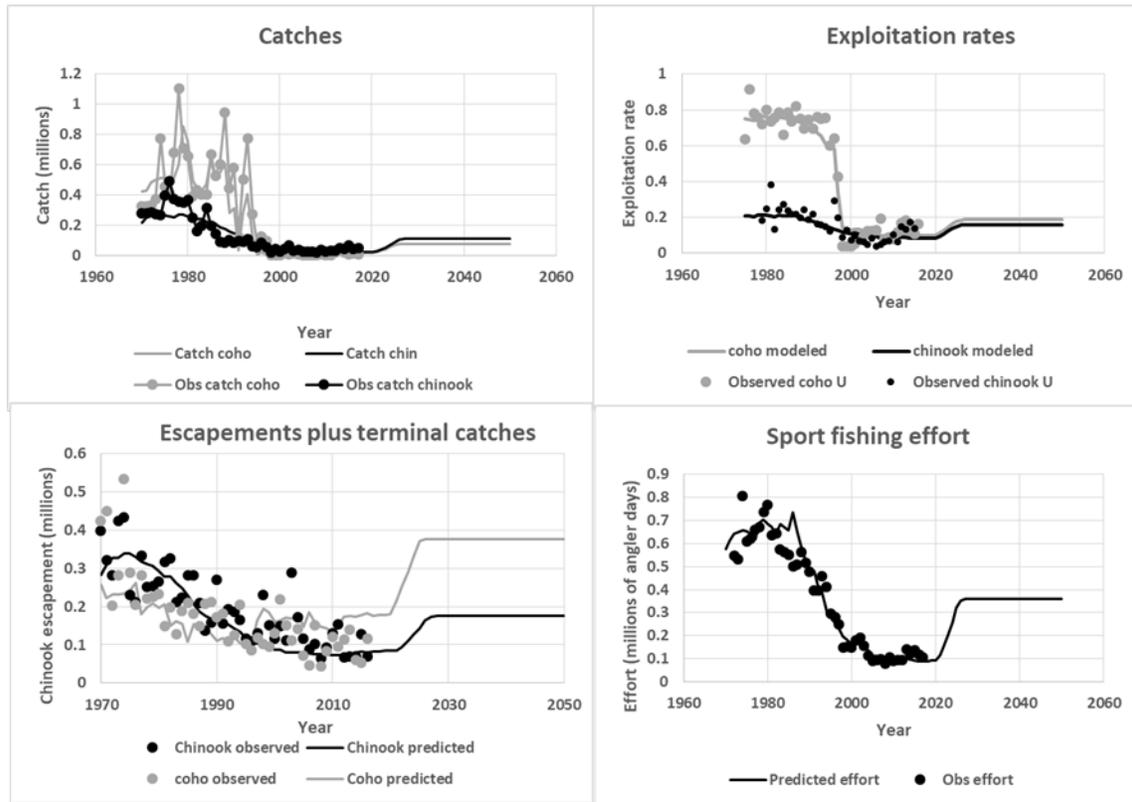

Figure 6. Predicted and observed indicators of change for the Georgia Strait sport fishery. Lines are model predictions starting in 1970 and projected forward after 2020 with a 50% reduction in seal abundance. Dots are observed values. From Georgia Strait sport model.xlsx. Catches and efforts from DFO recreational catch surveys and Argue et al. (1983), Exploitation rates from coded wire tag analyses by DFO and Pacific Salmon Commission, escapements from DFO NUSEDS database corrected for missing information on many streams using mean of streams actually visited each year.

The spreadsheet format makes it simple to include a bit of extra accounting for Chinook salmon to keep track of abundances by ocean ages (2,3, and 4), as these are affected by exploitation rate, natural mortality, and maturation patterns; the details of that accounting need not concern us here, and predictions are insensitive to those details in any case. One thing that is difficult to incorporate in the predictions is possible future variation and/or progressive change in the proportion of Coho rearing outside the Strait (a cause of high variation in catches as noted above); we could include arbitrary or random patterns in residence proportion in the calculations, but such patterns would not affect the main predictions about response to culling (unless we assume an irreversible decline in residence proportion, which seems very unlikely based on historical data).

In close agreement with historical data, the four assumptions above predict a weakly convex (downward bending) trade off relationship between seal abundance and sport fishing effort (Figure 7). The convex pattern implies that relatively modest reductions in seal abundance could

lead to substantial increases in fishing effort, with nearly historical effort levels predicted to result from reduction off the seal population by about 50%. To hold the seal population at around 20,000-25000, the logistic seal population model (Fig. 1) predicts that an annual harvest/cull of around 2,000 seals would be required.
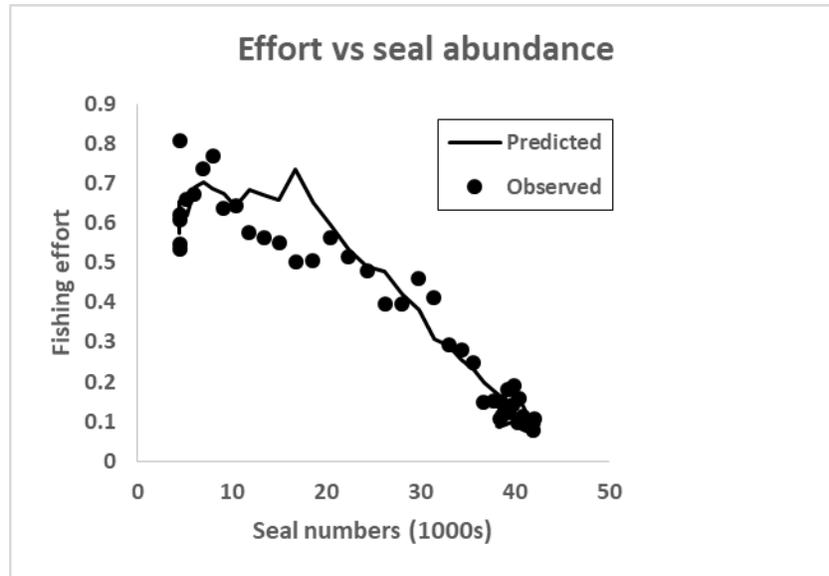

Figure 7. Modeled and observed trade off relationship between seal abundance and fishing effort.

It would not likely be practical or acceptable to simply reduce the seal population in one year to any moderate target abundance like 20,000. But if a harvest control rule (relationship between annual harvest and seal population size) were applied with the annual harvest being the surplus population size above the target level but capped an annual removal of say 4,000 seals, the model predicts a relatively rapid recovery of sport fishing effort (Fig. 6). There are obviously a lot of control rule options with alternative target seal abundances and limits on annual take. But the general pattern of response is predicted to be similar for all such options to the pattern shown in Fig. 6, just differing in how long the effort recovery is predicted to take and how high the eventual effort level.

Discussion

There is certainly no clear scientific consensus about the merits and risks of marine mammal harvesting or culling. On the one hand, there have been warnings from modeling studies about possible unintended, negative ecological impacts of harvesting top predators (Yodzis, 2001). On the other hand, there has also been strong scientific advocacy, again based mainly on modeling, for so-called "balanced harvesting" (Garcia et al. 2012) as an approach to marine ecosystem harvest management, where many species (and sizes) of organisms are harvested at rates proportional to their productivities. Tests of seal culling impact in Georgia Strait ecosystem models with structure similar to those used by Yodzis and Garcia et al. (Martell et al. 2001, Preikshot et al. 2013) have not revealed any unintended negative effects, but those models are by no means infallible.

Before Europeans brought disease and settlement to the Georgia Strait region, there was a large First Nations population that almost certainly harvested substantial numbers of seals for food and

other purposes such as skin bags for oolichan grease transport (Mckecknie and Wigen, 2011). So it is really difficult to argue that the current situation is in any way "natural", or that it would be best to simply let Nature take its course to establish some unprecedented new equilibrium, or that at least First Nations people should not be allowed to assert traditional rights to harvest of seals.

It remains possible that the observed correlation between juvenile salmon mortality rates and seal abundance is either (1) a spurious or coincidental one, or (2) symptomatic of other changes in the system that have made the juveniles more susceptible to predation, e.g. progressive spread of diseases that have weakened the fish, stress caused by growing contaminant concentrations, or decreases in effective food availability that have forced the fish to spend more time at risk to predation in order to achieve normal growth and maturation. Walters and Christensen (2019) show that the data can be explained by assuming linear increase in vulnerability to predation with increasing temperature, with the vulnerable fish subject to high mortality rates when not consumed by seals. Indeed, there are correlated patterns in deviations from the regression predictions in Fig. 3 that indicate other factors besides predation have influenced mortality rates over some time periods. Mortality rates of hatchery Coho have continued to increase well after the seal population stopped growing, again indicating that something beyond seal predation is continuing to have growing impact on at least those hatchery fish. Although not documented through scientific sampling, there is good anecdotal evidence from sport fishing that vertical distributions of Chinook and Coho have changed dramatically over the years. In the 1970s, we caught almost all Coho using surface (<10m) gears like bucktail flies, and it was rare to fish deeper than 30m with downriggers. But during the 1980s, surface gear stopped working and even Coho were typically caught at depths of 10-20m. In recent years, the most common recommended depths for downrigger trolling are well over 30m, up to even 60m. These depth changes could be symptomatic of fish trying to avoid predation risk, but could also be due to persistent and continuing changes in the basic trophic structure of food availability.

This basic uncertainty about causes of the decline means that any seal culling program should be treated as an adaptive management experiment, to be closely monitored and quickly abandoned if survival responses are not seen within a few years. Fortunately, the basic monitoring programs needed for an experimental approach are already well established (CWT tagging for survival, sport effort and catch monitoring, seal census protocols) and will likely continue to be funded. So there is no good scientific reason not to proceed with a before-after treatment comparison involving reduction in the most likely agent of increasing mortality based, especially considering that the predicted response is quite dramatic (see Fig. 6) and would not likely be confused with other causes of variation.

If we are correct in our basic arguments, there is very little chance that the Georgia Strait decline will reverse itself naturally without a strong and doubtless controversial policy intervention in the form of seal harvesting. This option has not yet been exposed for broad debate among stakeholders who care about the future of the Georgia Strait ecosystem, but it is past time to begin that debate.

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