Fisheries and Oceans Canada | Pêches et Océans Canada
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Canadian Science Advisory Secretariat (CSAS)

Research Document 2025/069

National Capital Region

Stock Assessment of Pacific Harbour Seals (*Phoca vitulina richardii*) in Canada in 2019

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November 2025

Canada

Foreword

This series documents the scientific basis for the evaluation of aquatic resources and ecosystems in Canada. As such, it addresses the issues of the day in the time frames required and the documents it contains are not intended as definitive statements on the subjects addressed but rather as progress reports on ongoing investigations.

Published by:

Fisheries and Oceans Canada
Canadian Science Advisory Secretariat
200 Kent Street
Ottawa ON K1A 0E6

http://www.dfo-mpo.gc.ca/csas-sccs/
DFO.CSAS-SCAS.MPO@dfo-mpo.gc.ca

ISSN 1919-5044
ISBN 978-0-660-79194-4 Cat. No. Fs70-5/2025-069E-PDF

Correct citation for this publication:

Tucker, S., Majewski, S., Nordstrom, C., Trzcinski, M.K., and Doniol-Valcroze, T. 2025. Stock Assessment of Pacific Harbour Seals (*Phoca vitulina richardii*) in Canada in 2019. DFO Can. Sci. Advis. Sec. Res. Doc. 2025/069. v + 82 p.

Aussi disponible en français :

*Tucker, S., Majewski, S., Nordstrom, C., Trzcinski, M.K. et Doniol-Valcroze, T. 2025. Évaluation des stocks de phoque commun du Pacifique (*Phoca vitulina richardsi*) au Canada en 2019. Secr. can. des avis sci. du MPO. Doc. de rech. 2025/069. v + 92 p.*

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ABSTRACT

Standardized aerial surveys were conducted between 2015-2019 to assess the abundance of Harbour Seals in British Columbia (BC). Approximately 90% of the entire coastline was covered using fixed-wing aircraft to count seals hauled-out on land during specific low-tide windows. Five years were required to survey all regions of BC, and thus, this assessment represents a compilation of surveys. Thirty-two satellite transmitters were deployed on adult and juvenile seals between 2019-2021 to estimate the proportion hauled-out, and to calculate a correction factor for animals at-sea and not present at the time of the surveys. An estimate of 78.5% of the seals were hauled-out during the survey period. This marks a substantial change from the last derivation from the early 1990's, when 62% of the seals were hauled-out. After applying the most recent correction factor, adjusting estimates for survey coverage, and summing regional abundance estimates in the year surveyed, 84,500 (95% CI 81,160 to 87,970) harbour seals were estimated in BC in 2015 – 2019. Projecting all regional trends to 2019 yielded a total estimate of 86,000 (95% CI 74,750 to 98,990) harbour seals in BC waters. Potential Biological Removal (PBR) estimated at 4,895 seals in 2019. While regional PBR allocations were estimated, they were deemed problematic as they may lead to local depletions. Correcting past surveys for uncovered areas resulted in an updated estimate for 2003-2008 of 112,400 (95% CI 108,000-117,000) seals, which is similar to the initial estimate of 105,000 seals (95% CI of 90,900-118,900). Given the uncertainty in the regional estimates, the stock in 2015-2019 is considered either stable or in slight decline relative to the 2003-2008 assessment. Abundance, density and trends varied regionally.

1. INTRODUCTION

The Pacific Harbour Seal (*Phoca vitulina richardsi*) is the most ubiquitous pinniped species in the Northeast Pacific, found throughout coastal and estuarine waters of British Columbia (BC) (Olesiuk 2010; Muto et al. 2021; Caretta et al. 2021). Their impacts on ecosystem dynamics are thought to be large due to their high abundance and position in the food web as a meso-predator (Chasco et al. 2017a & 2017b, Nelson et al. 2019 & 2021). The distribution and behaviour of harbour seals appears to be linked to prey availability (Harvey 1987, Thomas et al. 2011), predation pressures from killer whales (*Orcinus orca*) and shore-based predators (Nordstrom 2002, London et al. 2012), as well as human disturbance (Jansen et al. 2015).

There is ongoing interest in the role of harbour seals both as key predators of fish including herring, hake, and salmon (Olesiuk et al.1990a, Cottrell 1995, Lance et al. 2012, Li et al., 2010, Priekshot et al. 2013), and as a prey species critical to the recovery of transient killer whales, an ecotype present in BC that feeds exclusively on marine mammals (Ford et al. 2013). The ongoing assessment of the distribution and abundance of key prey species of transient killer whales has been identified as an important recovery objective (DFO 2007). In BC and Washington state, United States, perceived impacts of seals on salmon have renewed calls for regional population control measures (Trites and Rosen 2019, Trzcinski 2020, Nelson et al. 2023). Furthermore, increased interest in Food, Social or Ceremonial (FSC) harvests in BC are compelling managers to consider regional harvest allocations without complete and up-to-date information on stock structure or status.

Genetic analysis indicates that there are at least three populations of harbour seals in the Pacific (Burg et al. 1999; Huber et al. 2010 & 2012):

1. Japan, Russia, Alaska, and northern British Columbia;
2. Southern British Columbia and Puget Sound, Washington; and,
3. The outer coasts of Washington, Oregon, and California.

The northern British Columbia population would include the Northern Mainland Coast (NMC) and Haida Gwaii (HG) regions in this study, while the southern BC populations would fall into the five other regions (Figure 1). However, harbour seals in BC are managed as one stock, so we provide an abundance estimate for all of BC.

Historical reconstructions indicate the stock in BC was depleted by a period of harvesting during 1879-1914 to approximately 20,000 animals, and subsequently maintained below natural levels by predator control programs until the early 1960s (Olesiuk 2010). Already depleted, this stock underwent a second period of intense harvesting during 1962-1968 and was further reduced to approximately 10,000 individuals (Olesiuk 2010). Starting in 1970, harbour seals and all pinnipeds in Canada received statutory protections, and continue to be protected and managed under the Marine Mammal Regulations, pursuant to the *Fisheries Act*.

Fisheries and Oceans Canada (DFO) has conducted systematic surveys of harbour seals in BC since the mid-1960s, by counting harbour seals hauled-out on land during peak diurnal haul-out periods (typically low tide), during or just after the pupping season. These aerial censuses were conducted periodically throughout the 1970s, and more regularly since the early 1980s using standardized approaches. Olesiuk et al. (1990b) analyzed survey data collected up to 1988, and concluded that harbour seal abundance in BC had been increasing at a rate of about 12.5% per year. Using a rudimentary correction factor based on the variability among replicate surveys, it was estimated that harbour seal abundance had increased from ~9,000-10,500 in the early 1970s to about 75,000-88,000 in 1988. From a series of flights conducted between 2003-2008, which surveyed large portions of all regions of the BC coast, the first BC-wide estimate of harbour seal abundance was 105,000 (95% confidence interval (CI) of 90,900 to 118,900), which is similar to the estimates of historical abundances prior to large removal from hunting and bounties (Olesiuk 2010).

The Strait of Georgia (SOG) has the most extensive time-series of population estimates in BC due to the combination of a high density of animals (an average of 13.1 seals per km of coastline vs. 2.7 seals in other areas of the coast; Olesiuk 2010), the presence of culturally and commercially significant fish stocks (e.g. herring, salmon), and the relative ease in which the SOG can be surveyed. In 2008, the abundance of harbour seals in the SOG was estimated at 39,100 (95% CI 33,200 to 45,000) seals, representing approximately 37% of the BC stock (Olesiuk 2010). Abundance in this region remained unchanged when surveyed again in 2014, at 39,300 seals (95% CI = 33,400 to 45,200) (Majewski and Ellis 2021). Other coastal regions of BC have been surveyed much less frequently, and the abundance in these regions has not been re-assessed since the last coast wide assessment in 2010, at which time they showed either increasing or stable trends (Olesiuk 2010).

Surveys are flown near low tide during the peak pupping season to maximize the number of animals hauled-out and available to be counted (Olesiuk 2010). There are a limited number of survey opportunities with such specific survey conditions during a season (the survey window). Consequently, it takes several breeding seasons to cover an area as expansive as the BC coast, which has approximately 40,000 km of shoreline (Canadian Hydrographic Service; Data products and surveys; charts.gc.ca). Thus, a coast wide assessment represents a compilation of surveys over ~ 5 years (Olesiuk 2010; this assessment), and consequently, is not an all-inclusive ‘snapshot’ of the BC stock in time. The BC coast can be broadly divided into seven regions (Figure 1; Zacharias et al. 1998; Rubidge et al. 2016), where one or two regions (or parts thereof) are covered in a survey year. Due to weather, logistical and financial constraints, it has not always been possible to cover the entire coastline of a region. Therefore, estimates of abundance also require a correction for un-surveyed areas.

This compilation is made with the assumption that the population size has not dramatically increased or decreased while other regions are being surveyed, and that movement in and out of any region is approximately equal. Harbour seals have long-term site fidelity and high philopatry to natal areas, as demonstrated by tagging, branding and genetic studies (Härkönen and Harding 2001; Lowry et al. 2001; Steingass et al. 2019). While individuals may undertake occasional longer forays of up to several hundreds of kilometers, they typically return to their natal area for the summer breeding period, and between foraging trips or bouts (Pitcher et al. 1981; Yochem et al. 1987; Bjørge et al. 1995; Suryan & Harvey 1998; Härkönen and Harding 2001; Lesage et al. 2004; Cunningham et al. 2009; Dietz et al. 2013; Cordes and Thompson 2015). High site fidelity along with the observation that harbour seal population changes occur on the scale of decades without dramatic changes from year to year (Olesiuk et al. 2010; Muto et al. 2020; Carretta et al. 2021) supports the validity of a BC-wide population estimate compiled from surveys conducted over 5 years.

To estimate abundance, survey counts must also be adjusted for the proportion of animals that were at sea at the time of the survey, and therefore unobserved. Counts are typically corrected by applying a haul-out correction factor (CF), which is calculated by deploying satellite transmitters and using the telemetry data on numbers hauled-out (with wet/dry sensors). The last deployments of satellite tags on BC harbour seals were undertaken in the early 1990’s, and Olesiuk (2010) found that 62% of animals were hauled-out and available to be counted. In support of this assessment, we updated correction factors by deploying satellite tag over multiple years. Typically a CF is a simple estimate derived from the proportion of tagged animals which are hauled-out during the survey window (Olesiuk 2010). Here, we expand upon these methods to account for correlated behaviour among individuals in haul-out timing (Doniol-Valcroze et al. 2016).

DFO Management requested science advice on the current status of harbour seals in BC waters. Specifically DFO-Science was requested to provide:

1. An updated estimate of the current size and distribution of harbour seals in Canadian Pacific waters;
2. An estimate of the Potential Biological Removal (PBR); and,
3. If possible and appropriate, to consider PBR regional/local allocations (see Figure 1) while taking into consideration spatial variability in population abundance and density.

We present results from surveys undertaken between 2015 and 2019 throughout the BC coast, and correct for the proportion of animals that were at sea at the time of the survey using recent telemetry data. We also improved the methods used to estimate survey coverage by bringing them into a contemporary GIS framework. In the Methods, we outline where there is divergence from previous assessments and their estimates, describing and contrasting approaches under "Previous Assessments" and "New Assessment Framework." We also provide the first BC-wide and Regional estimates of trends in abundance over time, as this is an important component in providing PBR.

2. METHODS

2.1. STUDY AREA AND CENSUS TECHNIQUES

The BC coast was divided into seven regions, each of which were subdivided into 1 to 7 subregions: Strait of Georgia (SOG), West Coast Vancouver Island (WCVI), Queen Charlotte Strait (QCS), Discovery Passage & Jervis Inlet (DPASS), Haida Gwaii (HG), central mainland coast (CMC), and northern mainland coast (NMC) (Figure 1; Olesiuk, 2010, Majewski and Ellis 2021). Region and subregion definitions have been retained to facilitate comparison with previous surveys (Olesiuk 1999; 2010, Majewski and Ellis 2021), but have little demographic foundation as current data does not support this level of harbour seal population substructure (Muto et al. 2020). The regions and subregions correspond to Fisheries Management Areas, although not perfectly, and mainly reflect harbour seal survey logistics (i.e. what could be reasonably flown in a day). The regional delineations do, however, mirror 'ecosections' as defined by physical, oceanographic, and biological characteristics, and are used in the Marine Ecological Classification of British Columbia (Zacharias et al. 1998; Rubidge et al. 2016).

Surveys were undertaken in 2015-2019 to estimate stock status (Figure 2, Table 1). Survey flights were conducted following standardized census methods described by Olesiuk (1999, 2010). Survey protocols select for conditions when the maximum numbers of seals are expected to be hauled-out, and hence surveys 1) were conducted during or after the peak pupping season, and 2) were timed to coincide with low tides (0 to 1.5 m above datum) that occurred between 08:30 and 11:30 Pacific Daylight Time (PDT), and 3) began approximately two hours prior to the lower daily low tide, and ended approximately two hours after low tide (Figure 3).

Pupping in southern parts of the coast occurs in early July to mid-August (peak July 27; Bigg 1969b). The 2015-2019 surveys in the SOG, WCVI, QCS and DPASS were flown between 03 August - 09 September, in keeping with previous assessments (Olesiuk 2010). Pupping occurs earlier in the northern regions of BC, from mid-May to early July (peak June 10; Bigg 1969a). Therefore, survey timing was adjusted accordingly in NMC, CMC and HG regions, which were flown primarily in June/July in both this and previous surveys (Olesiuk 2010, Table 1). Flights were cancelled during inclement weather (i.e. rough seas, high winds or heavy precipitation) as seals appear to be less likely to haul-out under such conditions (Olesiuk 2010). Surveys were also cancelled when thick fog and/or smoke affected visibility.

We followed survey protocols documented in detail in Olesiuk (2010) and Majewski and Ellis (2021). Briefly, aerial surveys were conducted from a Cessna 180 flown at an airspeed of 125 km·hr-1 and an altitude of 150-200 m. Shorelines were followed and all islands circumnavigated. As in past surveys, we conducted a detailed search of the entire survey area, checking all known haul-out sites, with 1-3 observers scanning (usually with the aid 8x42 binoculars) for new haul-out sites. Flight track lines were recorded at 1 sec intervals, as a precise record of survey coverage. Visual counts were made of small groups (< 3 ind.) of hauled-out animals. Larger groups were photographed with a hand-held Nikon D810 or D5 camera equipped with an f2.8 70-200 mm Nikon lens. From 2017 onwards, overview photographs were taken with a second camera equipped with a wide-angle lens to support photographic reconstruction of complex sites. Digital images were shot in JPEG format, and imported into Adobe Photoshop for analysis. Contrast and brightness levels were adjusted if necessary, counting areas delineated using the brush tool and seals tallied (differentiating swimmers vs. animals that were hauled-out) using the 'count' feature and custom actions in Photoshop. Photos were geotagged to track lines based on time to confirm haul-out locations, and to compare survey tracks and haul-out locations to previous surveys. Twenty-two percent of the haul-outs (n=446) were counted twice by different observers, independently choosing frames to count for each haul-out, to examine inter-observer variation associated with photo counts.

2.2. SURVEY COVERAGE

A BC coast wide population estimate requires a final correction for the shoreline not surveyed. As in Olesiuk (2010) and Majewski and Ellis (2021), counts were standardized for the proportion of the geographic area covered by the survey (see section 2.2.1 below), but used a new method for calculating survey coverage (see section 2.2.2 below).

Due to no-fly zones around airports and military zones, and highly developed/urbanized areas (e.g., parts of Burrard Inlet and Indian Arm), there are small sections of the coast within the SOG that have never been surveyed. In keeping with previous assessments, no adjustments for these no-fly zones were made in the current assessment (Olesiuk 2010; Majewski and Ellis 2021). It was assumed that these areas did not have many harbour seals.

2.2.1. Previous Assessment

Counts in the previous assessment (Olesiuk 2010) were corrected:

1. For the proportion of known haul-out sites covered within a subregion; and,
2. For un-surveyed shoreline by applying seal densities (seals/km) from covered areas within the subregion to the un-surveyed shoreline length (km).

The SOG was treated differently than other regions of the coast. In the SOG, adjustments for the proportion of haul-out sites covered were typically minor because only a few sites were missed. However, there was no adjustments made for the un-surveyed shoreline which could be substantial for select surveys (Olesiuk 2010; Majewski & Ellis 2021). Similar adjustments were larger for the other regions of the BC coast, as typically a smaller proportion of the known haul-out sites and/or coastline was surveyed. It is unclear exactly how the proportion of haul-out sites covered within a subregion was derived in previous assessments (Olesiuk 2010; Majewski &

Ellis 2021). We presume that it was the proportion of haul-out sites surveyed versus the total number of known haul-out sites, where sites not surveyed were indicated by non-significant ("ns"). However, the proportions reported in Olesiuk (2010, see Appendix II) cannot be reproduced, as some haul-out sites by year combinations were left blank, and in many cases, the reported proportion surveyed was very different from the ratio of surveyed haul-out sites to the total of the subregion. Our estimate of the proportion of haul-out sites surveyed was consistent with the values in Majewski & Ellis (2021) with some small variation.

Olesiuk (2010) adjusted for the un-surveyed shoreline, by standardizing counts to the shoreline lengths in DFO Pacific fisheries management areas (PFMA). However, a close examination in GIS shows that while this layer delineates boundaries between DFO Pacific fisheries management areas, it is an incomplete representation of the coastline per se (see Appendix A for an example), and results in an inaccurate determination of shoreline length of the study regions and the coast as a whole. The PFMA GIS shoreline length is 27,200 km but the actual length of the BC coast is 39,000 km (Canadian Hydrographic Service; Data products and surveys) with different shoreline totals by region / PFMA area. In Olesiuk (2010), the length of area surveyed (i.e., flight track lines) was standardized to this total shoreline length of 27,200 km with an estimated ~4,800 km un-surveyed between 2003 and 2008 in the regions outside of the SOG (see Olesiuk 2010, Table 6). Using our more precise estimate of shoreline length, the number of kms surveyed was re-estimated across the entire dataset, and the estimate of un-surveyed shoreline from Olesiuk (2010) revised to 11,000 km. Subsequently, the number of seals missed in all un-surveyed areas was re-examined for these survey years.

2.2.2. New Assessment Framework

In order to assess trends in abundance over time, an estimate of survey coverage is required for all surveys. Survey track lines were used when available to estimate the proportion of the geographic area covered by a survey. The lack of GPS track data for historical surveys (i.e., prior to 2003) made it challenging to estimate the amount of shoreline covered for each survey year. A list of visited haul-out sites was used for earlier years without track lines to derive a GIS-based method and reconstruct survey tracks by linking the haul-out sites recorded as surveyed.

2.2.2.1. Surveys with GPS tracks (2003-2019)

The total amount of shoreline surveyed was estimated using track lines generated by portable GPS receivers aboard the aircraft from 2003 onward. Track lines were analyzed using R software (v4.1 2021, The R Foundation for Statistical Computing) and the sf spatial package (Pebesma 2018). The Canadian Hydrographic Service low water polyline shapefile was used as a base map. Tracks were merged for each year and subregion combination to avoid duplicating coverage when track segments from the same survey overlapped (e.g., tracks intersect when circling over haul-outs and segments of flight paths can be repeated when large subregions require multiple days to complete). This joined survey track was buffered to 500 m, and then intersected with the low water layer. The length of shoreline surveyed (clipped shoreline from the overlapped track), the total shoreline length, and the estimated percent coverage was calculated for each subregion and survey year. Coverage for a specific subregion/year was classified as 'wide' or 'discrete' post-hoc based on the survey path relative to the extent of the subregion. Wide ranging surveys typically covered >70% of all available shoreline, though not always. Some 'wide' surveys sacrificed covering shoreline in detail in some sections to survey further afield due to localized bad weather or time/tide constraints resulting in a lower overall percent coverage. Discrete surveys targeted specific locales within a subregion and typically covered <33% of available shoreline. Exceptions arose for a handful of surveys with limited scope but which covered sections in detail resulting in a relatively high coverage.

All recorded GPS track lines were used to build and corroborate a new GIS framework (described below 2.2.2.2) for defining survey coverage (see above). It is worth noting that counts from some surveys with very low coverage, specifically those between 2009-2012, were not included in this or any previous assessment. These surveys were an attempt to “fill-in” portions of regions not covered and reported on in Olesiuk (2010), which included parts of HG, NMC, CMC, QCS and DP (Figure 2). However, the track lines were included here to contrast methods of estimating survey coverage.

2.2.2.2. Path reconstructions for surveys without GPS tracks (pre 2003)

Estimates for the amount of shoreline covered by surveys without tracks were generated using ArcGIS software and the NetworkX Python library by simulating a path between haul-outs which had been recorded as surveyed. The reconstructed paths were subsequently used to estimate past coverage. Briefly, lists of haul-outs visited and optional anchor points (e.g., logical start/end points) were generated for each subregion and survey year. These points were then ordered according to GPS timestamps from a modern survey track in the same area in order to provide a sequence. Subregion polygons were rasterized (‘1’ = land and ‘0’ = water), with land negatively buffered by 50 m. Then, a path was created using the haul-out sites and anchors as nodes and with the proximity to land values in the raster map as edge weights. The weighted graph ‘pulled’ the path to map cells at the interface of land and water as the path extended from node to the next nearest node according to the shortest distance provided by a nearest-neighbour calculation, and thus recreated a typical survey path along coastlines and around islands. The path was buffered, and the CHS low water polyline coastline was clipped using the buffered path, as was done for flight derived tracks. Reconstructed coverages were classified as ‘wide,’ or ‘discrete’ in a similar manner to surveys with GPS tracks. To confirm the approach, pathfinding estimates of subregion shoreline length surveyed (km) were compared to GPS survey track line estimates for surveys undertaken in 2003-2008.

2.3. COUNT ADJUSTMENTS FOR PUP TIMING

Observations in the SOG by Bigg (1969a) indicated that pupping was normally distributed over time with a mean pupping date of July 27 and standard deviation of 16 days (see Figure 2 and Equation 2 in Olesiuk 2010). In previous assessments, counts at haul-out sites were adjusted for survey timing relative to peak pupping, i.e., for pups that were yet to be born (Bigg 1969a, Bigg 1969b, Olesiuk 2010). In this assessment, surveys were generally conducted well after peak pupping, resulting in only a small number of individuals being added to the abundance estimate (less than 100 individuals). Olesiuk (2010) talked to seal hunters in the Skeena River subregion (in NMC) and estimated that pupping in northern BC occurred approximately 47 days earlier (10 June) than in southern regions. Earlier pupping in these regions was partially accounted for by conducting surveys earlier, but it was also necessary to adjust the counts by survey timing relative to 10 June to account for missed pups. It is unclear how Olesiuk (2010) adjusted for a cline in pupping between the northern and southern regions, in particular, for the Central Mainland Coast. Perhaps he used latitude to interpolated the peak pupping date between the two cumulative curve for pupping in the SOG and the Skeena River (see Figure 2, Equation 2, Olesiuk 2010). Rather than make assumptions about a cline in the timing of pupping, a peak pupping data of 27 July was used in the current assessment for southern regions (SOG, WCVI, QCS, DP), and 10 June northern regions (NMC, HG, CMC), where the proportion of pups born (P_{born}) is:

$$\mathrm{p_{born}} = \mathrm{N}(\mu, \sigma^2) \tag{eq.1}$$

where μ = 208 Julian day (southern regions), or 161 Julian day (northern regions), and σ = 16.1. Then the correction for unborn pups ($pupCf$) is:

$$\text{pupCf} = 1.25 - (\text{p}_{\text{born}} * 0.25)$$

(eq.2)

As Olesiuk (2010) noted, these adjustments to the counts are generally small as most surveys were conducted at or after peak pupping. Adjustments to pup counts were applied assuming no error, and no change in pupping phenology since the data collected in the 1960s, as there is currently no data to inform an alternative assumption.

2.4. HAUL-OUT CORRECTION FACTORS

To estimate abundance, survey counts must be adjusted for the proportion of animals that were at sea at the time of the survey, and therefore unobservable. This proportion was estimated for this and previous surveys by instrumenting harbour seals, and using telemetry data to calculate the proportion of seals hauled-out during survey conditions (Huber et al. 2001, Jeffries et al. 2003, Olesiuk 1999b, 2010). We maintained consistency with Olesiuk (1999b, 2010) by examining wet /dry sensor data during periods from 08:00 – 12:00 PDT when water levels matched those targeted by aerial surveys (0 – 1.65 m), which typically occur from May through September in the SOG. The current methodology relates haul-out behaviour to a specific set of environmental conditions which, assuming the relationship holds, allows it to be applied to surveys following the same parameters.

2.4.1. Previous Assessments

Olesiuk (1999b, 2010) estimated the proportion of seals hauled-out during surveys based on haul-out patterns documented using time-depth recorders (TDRs). Between 1990 and 1994, TDRs were deployed on and recovered from 33 animals at 10 haul-out sites in the SOG. Based on consistent haul-out patterns relative to timing of low tides, he generated haul-out response curves that varied in amplitude depending on the height and time of the low tide (see Figure 11 in Olesiuk 2010). Olesiuk then generated a haul-out response curve that approximated the tidal conditions during each survey flight to determine the correction factor for that day. The haul-out response curve was subsequently used to determine a correction to adjust each count during the survey flight based on the time it was made relative to low tide. The procedure has been termed the ‘variable CF’ approach (Majewski & Ellis 2021) as the CF varied by hour and day. Olesiuk (2010) then applied this variable CF approach to each survey within the SOG. An overall mean CF of 1.626 (Coefficient of Variation, CV=0.042) was derived from the SOG data. There is an absence of any comparable telemetry data for regions outside the SOG (deployments have been restricted to the SOG in BC to date). Therefore, the mean CF was applied to all count data *outside* the SOG on the assumption that haul-out behaviour was similar throughout the species range in BC, and that surveys were conducted under comparable conditions (Olesiuk 2010). As the SOG haul-out curves were not made available, this same mean CF was applied in the 2014 SOG assessment (Majewski & Ellis 2021). Majewski & Ellis (2021) contrasted the harbour seal abundance estimates in the SOG over the 1973-2008 time series when applying the mean CF or variable CF, and did not find significant differences. This is not an unexpected result given that survey design already takes into account the largest source of variation in haul-out behaviour (i.e., time of day relative to specific low tides). and surveys are timed to observe the greatest number of seals on haul-out sites.

2.4.2. New Assessment Framework

2.4.2.1. Instrumentation and data processing

To update the CF, 32 satellite transmitters (SPOT6-293A or SPLASH-297A, Wildlife Computers, WA, USA) were deployed on a mix of sex and age classes captured at four sites in the SOG from April to early August in 2019 – 2021 (Table 2, Figure 4). Given the health and safety protocols in place and travel restrictions due to the Covid-19 pandemic, harbour seal deployments did not commence at the same time each year, and were restricted to sites around Nanaimo where the researchers were based (Region SOG, Subarea: GULFISL, Figure 4). Each tag was paired with a floatation pack and a radio-transmitter (Advanced Telemetry Systems, MN, USA) to assist with instrument recovery. Units recorded time at 1-sec intervals and wet-dry status every 2 seconds. A UHF radio-receiver (Wildlife Computers MOTE®) was installed at Entrance Island (2019) or Snake Island (2020, 2021) to log data transmissions from tagged seals (when they were within range) in an effort to supplement data transmitted through the Argos satellite network.

Seals were captured individually on haul-outs using hoop-nets following a rapid approach from a 16 foot aluminum vessel. Seals were transferred to a custom made restraint board and processed on-site. Seals were manually restrained and the satellite transmitters were glued to the dorsal pelage mid-line between the shoulders with 5-minute epoxy resin (Devcon ®, MA, USA). Capture and animal handling techniques were completed under DFO Marine Mammal License (XMM1 2019) and approval by the DFO Pacific Region Animal Care Committee (Protocol #: 18-022A3R3).

Sensors recorded data continuously but devices were duty-cycled to pause transmissions to the Argos network when satellite coverage was poor or when seals had been hauled-out for > 2 hours. A minute was considered 'dry' if the wet/dry sensor was dry for at least 30 s in a minute; the tagged seal was considered hauled-out after 5 consecutive 'dry' minutes. The haul-out state was exited if the tag subsequently registered wet for at least 50 s in a minute. Hourly percent-dry summaries were downloaded for each seal for the duration of the deployment via the onboard data archive (when tags were recovered) or from the Wildlife Computers online data portal. The first 24 hours were subsequently removed to account for a potential impact on haul-out behaviour resulting from the tagging procedure.

To align seal haul-out behaviour (wet /dry) with tide state (height), tidal amplitudes for the study area as recorded by the Nanaimo Tide gauge (Station #7917) at 1-min resolution and encompassing the deployment periods (May 1st through October 15th) were obtained from the CHS data portal. Water levels were summarized by hour and the median was taken as an integrated measure of tide height (as experienced by the seals over that hour) to match the integrated hourly haul-out behaviour data. Median tide-heights for each hour were then merged with the harbour seal dataset and used to filter haul-out data during survey-like water levels (0-1.65 m) and times of day (08:00-12:00 PDT).

2.4.2.2. Correction factor calculations

Haul-out CFs based on telemetry studies have often estimated the average value of P (a proportion of an individual's time hauled-out). Doniol-Valcroze et al. (2016) show that this approach was flawed for walrus surveys because it cannot account for the large variability observed in counts, and does not model the error distribution appropriately. Instead, they suggested the proportion of animals hauled-out at any given time as being a more relevant metric. Overdispersion will appear if individuals are not independent from one another in their behaviour. Irrespective of the reason, this correlation results in higher variance than expected from a regular binomial distribution. The authors proposed that a more appropriate approach would be to model counts following a beta-binomial distribution as opposed to a uniform or a binomial distribution. This framework was applied here to determine the proportion of harbour seals hauled-out and its associated variance. The theoretical framework and a detailed explanation are provided in Appendix B.

The full telemetry dataset and a reduced dataset (with hours and water levels matching survey conditions) were used to assess the variance and the correlation of haul-out behaviour under a beta-binomial framework (Doniol-Valcroze et al. 2016, cf. Appendix B). The same calculations were applied to both sets of data to determine the overall variance when all tidal heights/times were considered, and to examine the change when narrowing the data to periods with survey-like conditions. The final CF was calculated using the reduced dataset.

Hourly percent-dry data were used to assess the proportion of time hauled-out and the timing of haul-out bouts for instrumented seals. Percent-dry timelines aggregated by hour were plotted against local time for each seal; cyclical haul-out patterns were typically evident (Figure 5) and coincided with the tidal cycle. Data exploration indicated a highly bi-modal distribution of time wet versus dry was for all seals and all years (Figure 6). Seals typically alternated from a fully wet (swimming) state to a fully dry (hauled-out) state with limited transition between behavioural states. To account for this behavioural pattern, the proportional dry data were converted to a binary behaviour dataset, with ≥ 50% dry indicating “hauled-out” and <50% indicating “not hauled,” for each hour and each tagged seal.

While all seals displayed the bi-modal haul-out pattern, the timing of haul-outs was notably different for 2 seals (Pv19SOG-04 & Pv21SOG-07) based on the satellite tag locations: one made extensive use of log booms, while the other departed the study area shortly after tagging to an area for which the corresponding local tide data is lacking. Both seals were removed from the correlation assessment and the subsequent CF determination. Notably, sample size changed throughout the deployment period between April and August each year. The number of seals tagged increased irregularly as clusters of seals were often instrumented weeks apart as a result of shifting tides and favourable capture conditions. Furthermore, the instrument packages were shed over eight weeks beginning in mid-August; therefore, the number of active tags decreased irregularly. Collectively, this resulted in a composite of different numbers of tagged individuals (2 – 13 seals) simultaneously in each year collecting data over the course of the study period (Figure 7).

Nonetheless, the primary goal remained to estimate the proportion of seals hauled-out at any given time and not to estimate the average time hauled-out for a given seal. Therefore, the proportion of seals hauled-out (p_ho) was first calculated by summing the number of hauled-out individuals for every hour of the deployment period where:

$$p\_ho(t0) = seal\_cnt\_ho \,/\, seal\_cnt\_data \qquad \text{(eq.3)}$$

and t0 = any given date-hour, seal_cnt_ho = the number of seals hauled-out, and seal_cnt_data = the number of seals with data.

The number of hours that were recorded concurrently by multiple tagged seals was then tabulated. Specifically, the number of date-hour occurrences (events) were counted for the varying numbers of seals over the deployments (i.e., number of unique hours covered by a single seal, by two seals, by three seals, etc.). The data was subsequently filtered to retain date-hours (events) covered by at least two seals, and grouped the proportions of seals hauled-out (as determined for each hour) by year. The mean (P_ho) and empirical variance of seals hauled-out was then calculated for each grouping (2 – 13 seals) as follows:

$$P\_ho(SealGroup0) = sum(p\_ho) / n$$

(eq.4)

where n = number of concurrent date-hour events, and SealGroup is a grouping of seals with concurrent data.

The overall mean proportion hauled-out (P) was summarized for the collection as a whole (pooled by year) to derive a CF (1/P) and calculate the associated error and 95% CI. These summaries were used to investigate the sensitivity of P to the minimum number of concurrent seals per-event in the analysis. While the expected binomial variance was calculated for groupings of seals (eq. 2 & eq. 3, section 2.1, Appendix B), it was not possible to calculate the correlation among seals (ρ), nor the measure of over-dispersion on which it relies to generate a proper CV for our CF, by using the data as-is. Because the formula for ρ is a function of the sample size N, it could not be applied directly to our dataset where N varies over time (range = 2 – 13). Instead, it was assumed that the dataset consisted of a mixture of 12 beta-binomial distributions (one for each "seal group" with sample size from 2 – 13). Then, by employing the overall mean as parameter P, the correlation ρ was estimated from the total variance in the dataset (combining all the tagged individuals) by weighting the relative contribution for each distribution (i.e., by the number of date-hour occurrences for each sample; see Appendix B Section 3.2 for additional details on the analytical framework).

2.5. SUMMARY OF CHANGES TO ASSESSMENT FRAMEWORK

To recap, two corrections were applied to the count data to generate an estimate of abundance:

1. Unborn pups; and,
2. The proportion of animals in the water (not hauled-out) at the time of the survey.

Subsequently, the abundance was adjusted (corrected) for the proportion of shoreline surveyed. The timing of the survey aimed at minimizing the first two corrections; the last correction was mostly unavoidable and ultimately a question of resources and circumstance.

The first coast wide assessment was based on surveys undertaken between 2003-2008, which estimated a total of 105,000 seals (Olesiuk 2010). The correction for the number of unborn pups was based on defined pupping phenology from 1969 (Bigg 1969a), and an assumed earlier timing for peak pupping in northern regions (Olesiuk 2010). The proportion of seals hauled-out was based on CF derived from satellite tagging in the SOG in the 1990's with a 'variable approach' used exclusively for the SOG and a mean CF applied to all other regions. In Olesiuk (2010), the proportion of area surveyed was based on two corrections. First, there was an accounting of known haul-out sites surveyed/not-surveyed. However, discrepancies were noted in reported proportions (Appendix II in Olesiuk 2010 versus our calculations). Second, shoreline density corrections were also applied to the lengths of shoreline which were not surveyed (Table 6 in Olesiuk 2010, and Section 2.2 in this report). These areas of un-surveyed coastline were corrected by extrapolating mean subarea densities of seals to those missed shoreline portions. However, Olesiuk (2010) scaled these estimates to a fisheries management area GIS layer with an underestimated coast wide shoreline length of ~27,000 km. In the SOG only, when an entire subarea was missed, abundance was interpolated using an exponential growth model.

Initially a simple replication of the prior coast wide assessment was attempted, but some inconsistencies were identified which required details and clarification that were unavailable, thus unresolvable. Therefore, the 2003-2008 abundance was re-estimated by applying a similar approach to the original (Olesiuk 2010) but with modified inputs. The same correction for the number of unborn pups was applied as in the past, but with a systematic use of the equation in

Bigg (1969a). The mean 1990's haul-out CF was used and applied to all areas of the coast, including the SOG. The proportion of haul-out sites surveyed was not used to correct for missed haul-out sites since this would be a double correction with the shoreline density corrections. Shoreline density corrections were used in the same manner as previously, but based on a GIS layer with a true coastline length of ~39,000 km. For the SOG, there was no interpolation for missed subareas using an exponential growth model, however, the number of unseen seals in these areas was estimated using mean seal density and the amount of shoreline missed. This same approach, with GIS-based estimates of survey coverage, was used for trend analysis where regional and coast wide abundance were estimated back to the 1960's.

Having reproduced the 2003-2008 abundance estimate, the same approach was applied to the new 2015-2019 assessment period. The only difference was that the updated mean CF was applied to all areas for the final estimate, as opposed to continuing to apply the CF derived in the 1990's. The new CF incorporated added variance due to the correlated haul-out behaviour of seals.

2.6. ESTIMATES OF ABUNDANCE

Harbour seal abundance (N) was estimated by:

$$N = C * pupCF * hauloutCF$$

(eq.5)

where *C* = count, *pupCF* = correction for timing relative to peak pupping, and *hauloutCF* = correction for animals not hauled-out (1 / proportion animals hauled-out). Abundance was estimated from counts within a subarea in a particular year. The estimates of shoreline coverage (section 2.2) was subsequently used to correct counts for the kilometers of shoreline un-surveyed in that subarea in that year. Seal densities per kilometer (*Dsubarea*) were calculated from the population estimates (*N*) and the estimated kilometers of shoreline surveyed (*KMsurveyed*) within each subarea as follows:

$$Dsubarea = \frac{N}{KMsurveyed}$$

(eq.6)

In cases where an entire subarea was not surveyed in a survey year, the average of the observed densities in that subregion was used on the preceding and following surveys (Table 3). This density estimate was applied to the kilometers of un-surveyed shoreline to arrive at a subarea abundance estimate where:

$$Nsubarea = N + Dsubarea * KMmissed$$

(eq.7)

All subarea abundances were subsequently summed to generate a regional estimate of abundance within a year. The regions outside of the SOG were much less frequently covered and it often took more than one year to survey them. Therefore, it was necessary to assume that counts across a group of years were representative of regional abundance. The survey year was attributed to the year in which the greatest number of haul-out sites in a subregion were surveyed ("attributed year"). This was based on an assumption that there were no large changes in abundance across survey years, and that there was equal movement in and out of a survey area. Using the appropriate inputs as defined above (Section 2.5), regional abundance was estimated from each available survey going back to 1966. Coast-wide abundance was also re-estimated for the 2003-2008 period by summing all regions.

In a similar manner, abundance for the 2015-2019 period was estimated. For the 2015-2019 surveys, abundance estimates were contrasted using the previous average CF (Majewski and Ellis 2021) and the newly derived CF from harbour seals tagged between 2019 – 2021. It was assumed that seal density in un-surveyed areas was similar to those observed in surveyed areas. To evaluate this assumption, a sensitivity analysis was run where assumed seal density in un-surveyed areas was changed by 20% while refitting trend models (Section 2.7).

Previous assessments estimated total variance in abundance as the combined variance from the adjusted counts (counts already adjusted for survey timing and timing of pupping) and the CF (number of animals at-sea), using the delta method (Olesiuk 2010; Majewski and Ellis 2021). As noted by Olesiuk (2010), counting error and variation in counts due to animals hauled-out are likely confounded. Inter-counter error was small and not incorporated in this assessment. Abundance estimates prior to the current assessment used variance estimate from Olesiuk's (2010; CV=0.042). For assessment of abundance from the 2015-2019 surveys, CVs were calculated on a subregion basis, and incorporated the additional uncertainty due to correlated haul-out behaviour (beta-binomial dispersion factor see Appendix B). Variances were summed to arrive at the regional and BC-wide estimates of variance.

2.7. TREND ANALYSIS

There were only occasional surveys of regions other than the SOG prior to the late 1980's, and coverage only exceeded ~50% of the shoreline in these regions in the 2000's (see Results Table 9). Trends in abundance in these regions was estimated using an BC-wide harbour seal abundance in 1966 reconstructed by Olesiuk (2010) by using data from pelts, hunting returns, and bounties (see his Figure 13). He estimated that the abundance declined from approximately 80,000 seals in ~1887 to approximately 10,000 seals by 1970. Surveys started in 1966 in four of the subregions of the SOG (GULF, BBAY, FRASERR, GULFISL), and 651 seals were counted. Using the average CF for animals at sea of 1.63, our estimate of the amount of shoreline uncovered, and a reasonable supposition of the density of seals in the three un-surveyed subregions (HOWESD, NWGULF, NEGULF; Table 3), an estimated ~2080 seals occurred in the SOG in 1966. This SOG estimate was subsequently subtracted from Olesiuk's (2010) BC-wide estimate of 10,000 seals to apportion the remainder abundance to the regions. This was done based on the proportion of shoreline in each region relative to the BC coastline outside of the SOG. Thus, estimated abundance were as follows: WCVI = 1320, QCS = 880, DP = 616, CMC = 1672, NMC = 2376, HG = 1056 harbour seals. These values became starting values in the current analysis of trends in abundance in these regions. Generalized additive models (GAM) were fit to the abundance estimates in each region, except for DP where a linear intercept only model (GLM) was fit, with the proportion of shoreline surveyed as weights. Quasi-Poisson errors were assumed when using the gam function of the mgcv R package. The predicted values from each region were summed (and associated variance) to arrive at BC-coastwide trends. For regions outside the SOG which were surveyed between 2015 and 2018, models were projected to 2019, and these model estimates, along with the 2019 predicted abundance estimate for the SOG, were summed to obtain an estimate of harbour seal abundance for the entire BC coast in 2019.

In addition to modelling regional trends in abundance, harbour seal abundance trends were estimated in the subregions of the SOG by fitting a theta-logistic model, as done previously in Olesiuk (2010) and Majewski and Ellis (2021), where:

$$N_{t+1} = N_t e^{r_m\left(1-\left(\frac{N_t}{K}\right)^{\theta}\right)} \tag{eq.8}$$

and N_t = harbour seal abundance at time t, r_m = the maximum rate of increase, K = carrying capacity, θ = the shape of density dependence. The initial starting parameters for the theta-logistic can be found in Table 4. The theta-logistic model was fitted to the data using maximum likelihood and the log-normal error distribution with the proportion of shoreline surveyed as weights (the percent haul-out sites surveyed was used as weights in Olesiuk 2010, and Majewski and Ellis 2021). The *mle2* function in the *bblme* package in R version 4.1.1 was used to fit these models.

2.8. POTENTIAL BIOLOGICAL REMOVAL (PBR)

While there was reasonable survey effort dating back to the 1970's in the SOG, survey effort has not been even across all areas of the BC coast. Moreover, there is little additional life history information (e.g. levels of mortality, reproduction, trends in mean age/sex composition) to provide insight into stock dynamics or trends, or inform a more detailed stock assessment model for Pacific harbour seals in BC. Therefore, the Potential Biological Removal (PBR) approach was applied to determine sustainable removals of seals, and was calculated as:

$$PBR = 0.5 * Rmax * F_R * Nmin \qquad \text{(eq.9)}$$

where R_{max} is the maximum rate of increase, which was set to the default of 12% for pinnipeds (Wade and Angliss 1997, NMFS 2016); F_R is a recovery factor (between 0.1 and 1), and N_{min} is the 20-percentile of the log-normal distribution of the most recent abundance estimate (Wade 1998). The F_R that is applied depends on our understanding of stock status (DFO 2018; their Table 4).

PBR was calculated for the coast wide population as a whole, based on the N_{min} from the abundance estimate projected to 2019. The recovery factor was selected based on interpretation of the overall trend following trend analysis (see Results).

Science was also asked to consider regional PBR allocations. Although the seven survey regions do not represent distinct populations, managing at a regional scale can help avoid local depletion. Instead of simply partitioning the coast wide PBR by the proportion of the population within each region, each region was treated independently. These regional PBRs were calculated using region-specific N_{min} in their respective survey year and region-specific recovery factors based on regional trends and uncertainty in abundance following trend analysis (section 2.7).

3. RESULTS

3.1. STUDY AREA AND CENSUS TECHNIQUES

A total of 61 flights were logged between 2015 and 2019 (Table 1). They covered all regions and subregions within BC, and covered a total of 34,789 km of the 39,564 km of the BC coastline, or 91% of the BC coast. This compares to 74% coverage for the 2003-2008 survey period. Regional coverage ranged from 80% in WCVI to 94% coverage in HG (Figure 2).

3.2. SURVEY COVERAGE

Survey coverage was reassessed for all surveys from 1966 to 2019 based on either survey track lines or a reconstruction of survey tracks.

3.2.1. Surveys with GPS tracks (2003-2019)

Seventy-nine survey coverage estimates were generated using GPS tracks from flights conducted between 2003–2019 (Figure 2, Table 5). These flights were primarily from surveys qualified as 'wide' ranging (n = 69) in both periods. Coverage was more variable among flights during the 2003–2008 interval with 8 of the 37 estimates classified as 'discrete' (22%), versus 2 of the 42 estimates (5%) for the period 2014–2019. For example, the NW coast of Haida Gwaii was widely surveyed in 2008 but a patchwork of sections could not be completed (Figure 2 top left panel). A separate follow-up survey in 2010 targeted select stretches of NW coastline and the northern coast to achieve complete coverage 'in composite' (Figure 2, top right panel). By way of contrast, the complete NW corner of Graham Island was flown in 2017 resulting in a more cohesive survey (Figure 2, bottom left panel).

3.2.2. Path reconstructions for surveys without GPS tracks (1966–2000)

Path reconstruction and coverage estimates were generated for 121 historical surveys without GPS tracks (1966–2000). In addition, path reconstructions were generated from 38 surveys from 2003–2008 which had GPS tracks in order to assess reconstruction performance (Tables 6; additional reconstructions for 2009-2012 included for completeness, but are not used in this assessment). The vast majority of reconstructed paths were classified as 'wide ranging' (85%) and flown in the SOG (n=93). Figure 8 (left panels) shows examples of GIS path reconstructions with estimated coverages for Howe Sound in 1988 and 2008. The estimated coverages from reconstructions ranged from 5 to 56% for 'discrete' surveys and from 40 to 98% for 'wide' surveys.

3.2.3. Assessing path reconstruction

Comparing the GPS track to reconstructed ones for a subset of 38 flights flown between 2003 and 2012 (Figure 9), indicated the two methods were highly correlated ($r^2 = 0.78$), but with a significant effect of survey type (wide ranging versus discrete; $F_{1,29} = 7.22$, $p = 0.012$). The correlation improved when excluding discrete surveys and considering only the wide ranging ones ($r^2 = 0.96$), which comprised the majority of path reconstructions for historical surveys without a recorded track line.

3.3. COUNTS

The unadjusted counts and fully-corrected counts for 2015-2019 are summarized by region and for BC as a whole in Table 7. Counts by haul-out site (Figure 10) are available as the 'Harbour Seal dataset' via the Open Government Data Portal.

3.3.1. Inter-reader errors

There was a high correlation between the total counts of two readers who counted harbour seals at 446 haul-out sites, which was not significantly different from the 1:1 line (Figure 11, $r^2 = 0.95$). Therefore, no correction was made for inter-reader errors.

3.4. HAUL-OUT CORRECTION FACTOR

A total of 20 females and 12 males (21 adults vs 11 juveniles) were equipped with satellite transmitters over the course of the study, providing 44,494 'seal hours' of data. Deployment duration ranged from 9–131 days although tags were typically carried for approximately 1–3 months (Q1-Q3 = 36 – 84 days). Of the 32 tags deployed, two seals left the study area and were not considered further in our analysis. Twenty-two of the 26 recovered instruments were SPLASH tags which provided access to full archives of detailed dive and haul-out data. The remaining 10 tags provided relayed data only: 6 units were SPOT tags deployed in 2019 (which do not store data onboard) and 4 units were unrecovered SPLASH tags deployed in 2020 or 2021 (Table 2). The amount of relayed-only haul-out data was greatly improved by combining the satellite data with data recorded by the local UHF receiver: upwards of 50% for some deployments. Notably, deployments with the most missing data were from 2019 when the receiving base station was situated at Entrance Island. Despite the data gaps, the proportion of time hauled was similar in 2019 for seals with transmitted-only data (70%) to those with archival data (73%).

3.4.1. Correction factor derivation

The proportion of seals hauled-out was initially explored using the full dataset with all hours when there were a minimum of 2 seals per date-hour (event) to quantify the correlation in haul-out behaviour among animals. Overall, the mean proportion of individuals hauled-out was approximately half (P = 31.7%, CI ± 0.75%, n = 7,295 h) that observed when the dataset was filtered to survey-like conditions (P = 76.1%, CI ± 2.6%, n = 302 events) (Figure 12). The unfiltered data had a wide and relatively flat distribution in the proportion of individuals concurrently hauled-out. Despite a modest peak when half the seals were hauled-out, they were most likely to all be at-sea at any given time as evidenced by the prominent spike when none were hauled-out (Figure 12).

Half the tagged seals were hauled-out relatively frequently in the filtered data, but in stark contrast, seals were most frequently observed all hauled-out simultaneously when limiting data to survey-like conditions. In this case, the distribution of the data was skewed to the right, but roughly centered on the mean proportion simultaneously hauled-out.

Given these patterns, the two datasets were explored by grouping data based on the number of seals concurrently providing information, and calculating binomial variance and beta-binomial variance for each sample size (eq. 2 & 3, Appendix B Figure 13). As expected, the variance decreased with the number of seals concurrently tagged. However, the variance was much higher than the expected binomial variance when using the unfiltered dataset, with a markedly reduced variance when limiting data to survey-like conditions.

The sample size groupings (number of tagged seals with data) allowed us to examine the error and confidence interval around the overall haul-out estimate (P) created from a composite number of seals varying from groupings of 2 to 7 seals (Table 8). Changing this threshold number of seals from 2 to 7 had a large effect on the data available (75% reduction in available data when requiring 7 seals), but resulted in only a slight increase in the mean proportion of individuals hauled-out (3%), slight widening of the confidence interval (2–4%), and a small reduction in the empirical variance (0.02). Standard errors were relatively small under all scenarios, but were smallest and stable when a minimum of 2–4 seals contributed data for a given hour (0.004, unfiltered dataset) or when a minimum of 3–4 seals contributed data for a given event (0.012, survey-like conditions).

Restricting the survey-like data to events with a minimum of 3 concurrent seals struck a balance between minimizing error (se = 0.012), lowering variance (0.035) and maximizing data (n = 236 events). As such, the mean proportion of animals hauled-out was estimated at 0.785, and the associated CF at 1.274 (1 / 0.785), which was then applied to the survey adjusted counts in the current assessment (2015 – 2019 survey data).

When using data restricted to n >= 3 seals per event, and accounting for mixing distributions, the correlation in haul-out behaviour was high for the unfiltered dataset with P = 0.385 (ρ = 0.291) and resulted in a 222% increase in variance. Filtering the data to survey-like conditions with P = 0.785 reduced the correlation to a negligible amount (ρ = 0.002) and resulted in a 9% increase in variance.

3.5. ESTIMATES OF ABUNDANCE

Correcting past surveys for uncovered areas resulted in an updated estimate for 2003-2008. When regional estimates were summed in each of the respective survey years 112,422 (95% CI 108,820-119,434) seals were estimated BC-wide. Projecting the model trends for each region to 2008 resulted in an estimate of 108,557 (95% CI 99,185-118,815) seals. These estimates were similar to the initial estimate (Olesiuk 2010) of 105,000 seals (95% CI of 90,900-118,900).

The sum of the regional estimates in each of the respective survey years (2015-2019) resulted (Table 7) in a BC-wide abundance estimate of 84,497 (95% CI 81,160 to 87,970) harbour seals. Projecting the model trends for each region to 2019 (Table 7), a slightly higher but statistically similar estimate of 86,015 (95% CI 74,745 to 98,983) was obtained. Given the uncertainty in the regional estimates, the stock in 2019 is either stable or has declined slightly relative to the 2003-2008 assessment. The use of the 1990's haul-out CF increased abundance estimate(s) by 22% (Figure 14).

Regional abundance estimates ranged from 3,280 (95% CI 2,779 to 3,872) seals in DP to 35,478 (95% CI 32,860 to 38,304) seals in the SOG (Table 7). Relative to the prior assessment period, abundance increased in WCVI but declined in all other regions (Table 9, Figure 14).

From the projected 2019 BC-wide abundance estimate, the SOG would contain 43% of the coast wide population with a mean density of 10.5 seals per km (Table 7). The next most important region would be HG (16%), WCVI (14%) and NMC (12%) with respective mean densities of 2.9, 2.1, and 1.0 seals per km. The remaining areas (QCS, DP, CNC) constituted 15% of the whole population with densities ranging from 0.6-1.6 seals per km.

3.6. TREND ANALYSIS

With the exception of WCVI and DP, the abundance of harbour seals was estimated to have increased in all regions of BC until the early or mid-2000's, then to have remained either stable or to have shown signs of decline (Figure 14). In contrast, seal abundance in WCVI continues to increase, with DP showing no trend. There is considerable uncertainty in the trends in abundance for most regions, given both the uncertainty in the starting 1966 estimate and the low frequency of surveys. Over the entire time series, correcting for un-surveyed shoreline resulted in a variable number of seals being added to the total, accounting for between 1% and 95% of the regional total (Table 9). Interpretation of whether the abundance is stable or possibly declining is therefore dependent on corrections for shoreline coverage. Modifying mean subarea densities by ± 20% resulted in only slight changes in regional trends in abundance, suggesting a low sensitivity to the assumed density of harbour seals in un-surveyed areas. That is, although abundance estimates changed, the regional trend in abundance did not change. This is shown in Figure 14, where trend lines for the sensitivity analysis (dashed) remained within the 95% CI of the best estimates of population trends, with the exception of the trend in QCS. While the pattern of fits were similar for this region, it was surveyed only three times within 10 years, and with poor coverage (>70% missed).

Regional trends (model predicted lines) were summed to estimate the trend in harbour seal abundance in BC. Based on the sum of regional trends, the BC-wide harbour seal abundance was estimated to have increased at a mean annual rate of 7.25% from 1966 to ~1985, at which point the rate of increase slowed. Abundance peaked at 115,000 individuals in 2002, and has remained stable or has slightly declined at a rate of 2.2% since 2010. It is worth noting that the model estimate of abundance in 1966 was higher than our inputs, which would have some influence on our understanding of rates of increase for the stock among regions and across BC. The sum of regional trends resulted in an estimated 13,415 seals in BC in 1966, as opposed to the rough model starting estimate of 10,000 seals in BC taken from Olesiuk's (2010) historical reconstruction.

The highest densities of seals are expected to be found in the SOG, and consequently, the SOG was the most intensively surveyed region between 1996 to 2019. To understand variation in rates of increase, density dependance, and carrying capacity, separate theta-logistic models were fitted to each subregion of the SOG. Harbour seals abundance increased in all subregions of the SOG from the 1970s to the mid-1990s after which it stabilized (Figure 15). Theta-logistic models fitted the data well until the mid-1990s, after which the abundance estimates were quite variable. There were notable declines in abundance in the early 2000s in the subregions of BBAY and FRASERR (refer to Figure 1 for SOG subregion delineations). At that time, the estimate abundance in HOWSED was the highest or second highest on record and above the theta-logistic fitted carrying capacity, possibly indicating movement between subregions. The overall abundance estimate of harbour seals in the SOG in 2019 was at or below the fitted carrying capacity. In general, parameters from theta-logistic models are highly correlated, and consequently difficult to estimate. The estimates of maximum rate of increase for each subregion ranged from 0.08 to 0.28 per year. Theta, which expresses how the growth rate of a population slows as abundance increases (shape parameter), ranged from 6.6 to 32.3 (Table 10).

3.7. POTENTIAL BIOLOGICAL REMOVAL (PBR)

Harbour seals are currently abundant and thought to be near historic levels, with increasing or stable trends throughout BC (Olesiuk 2010, this assessment). In accordance with guidelines for application of various levels of recovery factors for use in Canada (DFO 2018), they meet the criteria for a recovery factor, F_R=1. Based on the estimated coast wide abundance projected to 2019 and an N_{min} of 81,575, PBR for the BC stock is 4,895 seals.

The certainty in trends varied across regions, and consequently the values of F_R ranged from 0.1 to 1 (Table 11). The estimates for PBRs based on regional estimates and uncertainties ranged from 18 seals in DP to 2069 seals in the SOG (Table 11).

4. DISCUSSION

4.1. ABUNDANCE ESTIMATES AND TRENDS

The current assessment indicates a small decline in the abundance of harbour seals in BC waters relative to the 2008 estimate. However, uncertainty exists around estimates from the two periods, and pertains to the extension of the haul-out CF specific to the SOG to all regions, and the correction for shoreline coverage. Nevertheless, the current estimate is likely to be more reliable as survey coverage increased from ~70% of the BC shoreline in Olesiuk's (2010) assessment to 91% of the BC shoreline in the 2019 assessment (i.e., 24,774 km and 33,863 km, respectively). Harbour seal counts increased from 46,601 in Olesiuk (2010) to 59,123 individuals in this assessment. However, there was a decreasing rate of seal detection in the newly surveyed areas as the percent difference in coverage (27%) exceeded the difference in counts (21%). This supports the view that the harbour seal population in BC is stable or slightly declining.

In all regions except WCVI, the most recent abundance estimates were below the previous estimates with non-overlapping CI's; the recent estimate was higher than in 2008 for WCVI. There are varying degrees of uncertainty in the modelled fits in these regions (given lack of survey data and corrections for missed shoreline), but the models indicate an increasing abundance trend in WCVI, decreasing trends in QCS, CMC and HG, and stable trends in NMC. There is a significant difference in the 2019 abundance estimate in the SOG when applying the old versus the new CF, which also affects our perception of decline since the mid-2000s. The utilization of the new CF in areas outside the SOG for years 2015-2019 had little effect on abundance trends. However, one should be cautious interpreting abundances trends. Although the use of a new CF was warranted, it represents a substantial break in how the data are treated. The validity of the application of any CF back through time (whether 90's or 2020's) is uncertain. The abundance trends were contextualized by 'anchoring' them to a starting population of ~10,000 seals based on Olesiuk's (2010) historical reconstruction of the BC population. The sum of the regional model fits suggested instead an estimated starting population of ~13,400 seals. Without setting an initial population size the model estimated an unrealistic abundance of seals in the 1960's (>50,000 seals) (Olesiuk 2010). Since the initial population size was roughly approximated (Olesiuk 2010), we caution against the use of the modeled trend when calculating population rates of increase over the entire time series. A additional limitation to the analysis was the large gaps in survey coverage prior to the 2000's. While differences in survey coverage were accounted for in the current assessment, given the increased coverage, there is greater confidence in data acquired since the early 2000's and trends during that period.

With broadly similar survey effort in the SOG between 1992 and 2014, counts have been stable at ~23,000 seals, which translated into a mean estimated abundance of approximately 40,000 seals (Majewski and Ellis 2021). In 2019 counts increased for the first time since 1994, however, survey coverage was at a maximum, increasing from approximately 75% to 85% of surveyable areas of the SOG although all previous abundance estimates have been recast to incorporate differences in survey coverage. In the SOG, the 2019 abundance estimate was lower than in 2014, partially due to the updated CF, but CIs of the two estimates overlapped, suggesting the population remained unchanged or declined slightly over this period. Mean estimated abundance has bounced around in the SOG since the mid-1990's, but the Generalized Additive Model trend, fitted to abundance estimates in the SOG, suggests the population peaked in the early 2000's and has been declining since then. The theta-logistic model does not have the same flexibility as the GAM model, and therefore cannot fit the current observed decline in abundance. The theta-logistic trend in harbour seal abundance in the SOG is similar to what has been reported in the past (Olesiuk 2010, Majewski and Ellis 2021), with an increase from the 1970s to the mid-1990s, after which seal abundance stabilized and appeared to vary around carrying capacity, missing recent downward trends better captured by the Generalized Additive Model.

At the same time as there was an increase in seal counts in the SOG, an increase was also documented in the proportion of animals hauled-out during the survey window from 62% in the mid-1990's to 78.5% currently. It is unclear when or over what time period a shift in haul-out pattern occurred, and our results demonstrate how sensitive a harbour seal assessment is to the CF for animals at sea. We may have erroneously been under the impression that the population was stable, masked by an outdated CF, or perhaps this is a recent phenomenon. A future analysis of comparable deployments by the Washington Department of Fish and Wildlife in Puget Sound in 2014 and 2016 and a DFO archive from 2007-2010 in the SOG might provide some insight on when changes occurred. A primary hypothesis for the decline in seal abundance within the SOG would be the increased presence of transient killer whales (TKW; Shields et. al. 2018). Shields et al. (2018) estimated that in 2017 alone, approximately 240 transient killer whales spent 3,328 killer whale-days in the Salish Sea and consumed between 900 and 1,300 seals (mean 1,090 seals). Since all parameter estimates were conservative in the bioenergetics model, the authors concluded that it is likely the actual number of seals predated upon far exceeded this estimate (Shields et al. 2018). An increase in TKW predation might explain the declines in abundance observed in four of the seven regions. Increased predator avoidance due to increased predation risk is also a potential mechanism for the observed increase in the proportion of seals hauled-out between the 1990s and our study (2019 – 2021).

Trends in the estimated abundance for harbour seals in BC are similar to those observed elsewhere in the Pacific Northwest (Muto et al. 2020; Carretta et al. 2021). Both regional (State) and local (subregions/stocks) trends in US abundance have been monitored at various time intervals from California to Alaska since the 1970s, revealing diverse spatial patterns in stock trends. As of 2018, various trends (decreases, increases and no changes) have been documented across the 12 stocks identified in Alaska (Carretta et al. 2021). The last assessment in California in 2012 suggests that abundance had stabilized after a decline from a high in 2005 (Muto et al. 2020). While the last assessments for Washington (1999) and Oregon (2004) are dated, results mirror those for the SOG over the same period; an increase until the early 1990's then a stabilization in abundance (Muto et al. 2020).

4.2. SURVEY COVERAGE

The 2015-2019 surveys achieved the highest coverage of the BC coast to date, and the least amount of correction for un-surveyed areas thereof. This correction had a large impact on results, particularly for the historical dataset when large portions of the coast were not covered. For example, only 5,000 seals (6% of the total abundance) were added to the total in the current assessment, versus 31,000 seals in 2008 (28% of the total). While the regional trends in abundance were found to be insensitive to assumptions of seal densities for missed portions of the coast, these density corrections may require further refinement at smaller spatial scales to incorporate and reflect the range of complex features of the coast and habitat available to seals. The extensive coverage achieved during the 2015 – 2019 survey may not be possible in future surveys. In this context, a more thorough analysis of regional density patterns and trends might help guide and refine future survey work as well as abundance estimates for the historical dataset.

Prior to 2003, survey coverage was only accounted for by applying a correction for the proportion of haul-out sites within a subarea (Oleisuk 2010; Appendix 2). The 2008 BC-coastwide population assessment was the first time that a correction for un-surveyed areas using an estimate of shoreline density was applied. Correcting for survey coverage is important for estimating trends in abundance over time. The two methods for estimating effort in a region (track line and GIS pathfinding) proved to be comparable as they provided near identical estimates of shoreline coverage. Further refinement of this approach, including quantification of error from multiple runs, may be warranted in the future.

4.2.1. Surveys with GPS tracks (2003-2019)

During the 2003-2008 period, flight paths were recorded for two surveys in the SOG and for some of the first surveys for subregions in the CMC and NMC. Coverage during this period was typically more variable between years. In select subregions outside the SOG, it was common for a survey to encapsulate a large swath one year, and then to target a discrete (often previously unsurveyed) portion of the range in another. This was often a by-product of complex geography where a series of inlets and channels branch away from the main sounds. Partly owing to lessons gleaned from earlier GPS recordings, surveys since 2014 were completed more evenly in time with most areas surveyed in a single year. Two complete surveys of the SOG were achieved (Majewski and Ellis 2021; this assessment), along with synoptic surveys for subregions in all six other regions.

4.2.2. Path reconstructions for surveys without GPS tracks

The GIS pathfinding technique successfully recreated paths between haul-outs that mimicked aerial surveys that do not fly point-to-point, but rather follow the contours of the coastline at low tides. However, in order to recreate basic survey mechanics, a minimal set of anchors (pseudopoints) was added to the reconstruction for each subregion. Recreated paths must start from and end at a point, whereas most flights must transit in and out of an area and typically incorporate such travel into the survey. Select anchors were added to provide a logical start and end point for paths when appropriate, such as when a specific seaplane base was known to be used or when adjacent subregions were known to be completed in series.

Discrete surveys were more challenging to address. However they represented a small portion of the reconstructions used. Regardless of classification, shoreline coverage estimates generated from the GIS pathfinding technique should be considered minimum estimates of survey coverage. Despite issues with the method, GIS reconstructions were relied on only little in the worst performing subregions given the majority of these subregions had not been historically flown (i.e. prior to 2003), and therefore all surveys within their boundaries included GPS tracks.

4.3. HAUL-OUT CORRECTION FACTOR

Our instrument deployment sample size was similar to the effort in the 1990’s (n~30) upon which all previous BC abundance estimates have been based to date (Olesiuk 2010; Majewski and Ellis 2021). The updated deployments were more restricted in space in the SOG relative to the 1990’s (essentially one vs three different localities) due to highly restrictive travel policies under the DFO COVID-19 response. However, the sample size was more evenly balanced across size classes and between sexes (65% adults, 35% juveniles; 63% female, 37% male). Previous tagging in the 1990’s was slightly more biased towards adult animals (70% of the total).

The substantial increase noted in haul-out proportions since the 1990’s was obtained using a beta-binomial framework, focusing the derivation of CF and its error on a subset of the data when seals were more likely to be hauled-out. This strategy eliminated a large part of the extra variance, although calculating variances to apply the correct CV to the estimated CF was challenging as the data were composites of different number of seals, as detailed earlier. We found a strong correlation ($\rho = 0.291$) for the complete, unfiltered haul-out behaviour dataset. That is, over the course of an entire day, the probability of being hauled-out was highly correlated among individuals and produced an over dispersed estimate, with high variability (large CV), for the proportion of animals hauled-out. Limiting the data to only the survey conditions (4 hour morning periods during select low tides) reduced both the correlation in haul-out behaviour among conspecifics to a negligible value ($\rho = 0.002$), as well as the overdispersion. This seems counterintuitive, but when almost all individuals (78.5%) were hauled-out under these conditions, the subsequent behaviour of seals had no influence over the behaviour of other seals. It is likely that variances were lower because the majority of seals exhibited similar behaviour, supporting how effective the survey window was in optimizing survey counts.

In the current assessment, the abundance and variance in abundance were calculated for each subregion, and then summed across subregions to obtain regional and BC-wide estimates. The correlated haul-out behaviour of seals likely occurs at a spatial scale between a haul-out site and a subregion. Given this uncertainty, using the subregion as a unit of spatial stratification seemed like a reasonable compromise between calculating the variance in abundance at the haul-out site or the regional scale (see Appendix B.4).

From an ecological perspective, we would hypothesize that a shift in behaviour (greater proportion of seals hauled-out) may be due to two factors that may or may not be mutually exclusive: an increase in primary food sources (prey abundance), and an increase in primary predators (e.g., transient killer whales). Herring spawning biomass has reached near-historical highs in the SOG since the lows documented in the mid-2000's (Boldt et al. 2020), whereas TKW foraging time in the SOG has at least quadrupled over the last decade (Shields et al. 2018), and their population has also increased (Towers et al. 2019). TKW are sighted almost daily in the SOG in recent years, which was a rare occurrence in previous decades. Seals may be spending less time foraging, and/or more time evading predators, thereby increasing the proportion of animals hauled-out at any given time (regardless of the primary driver). Individual harbour seal foraging ranges also appear to be more restricted (< 10km; Nordstrom unpublished) relative to values reported in the literature (10's of kms; i.e. Suryan and Harvey 1998, Härkönen and Harding 2001, Cunningham et al. 2009, Peterson et al. 2012). Increases in other marine mammal species abundance may be increasing the complexity of interactions in the ecosystem as well, with increased potential for competitive interactions particularly around forage fish consumption. Sea lion presence and abundance has also increased over this period (Olesiuk 2018; DFO 2021; DFO 2023), as has the occurrence of cetaceans, particularly humpback whales and harbour porpoise (Wright et al. 2021). Conversely, there have been large declines in abundance of piscivorous Chinook salmon and Coho salmon (Zimmerman et al. 2015; Ruff et al. 2017).

4.4. PBR

PBR was estimated based on BC coast-wide abundance estimate in 2019. The underlying assumption is that seals within BC would all be considered to be from a single, relatively mobile biological population. However, harbour seals exhibit strong fidelity to natal areas (Härkönen and Harding 2001). Genetic differentiation among groups of seals in the northeast Pacific has been detected on the scale of a few hundred kilometers along a continuous distribution of over 15,000 km (Stanley et al. 1996, O'Corry-Crowe et al. 2001, Westlake and O'Corry-Crowe 2002) and further genetic diversity is emerging at smaller scales (Sutherland et al. 2024) suggesting subpopulations may need to be managed on a local scale. Not accounting for population substructure in allocating PBR could lead to local depletion. Different regional abundance trends outlined in this assessment indicate regional populations might recover at different rates, a statement further supported by the large variation in the rates of increase estimated from the theta-logistic models of seal abundance in the subregions of the SOG.

Other ways to consider the issue of spatial variation in harbour seal density in light of potential regional PBR allocations could be to adjust the input parameters of the PBR calculation, namely the recovery factor F_r, on a regional basis. This is analogous to Brandon's et al. (2017) proposal of a tiered hierarchy of data availability, from data-rich to data-poor. A PBR tier system allows the best available information to be used for each stock, recognizing the different types and levels of uncertainty that exists among stocks. This approach was applied in the current assessment. The different regions were treated as independent 'stocks' to derive region-specific PBRs based on respective trends and latest abundance estimates. Conversely, a harbour seal specific management strategy evaluation (MSE) with clearly defined management objectives might be undertaken. However, these would require re-visiting current harvest guidelines and in-depth spatial analysis, which are beyond the scope of this assessment.

5. CONCLUSIONS

The population size of harbour seals in BC is stable or slightly declining since the early 2000's. Standardized aerial counts have been found to provide a reproducible index of harbour seal abundance in BC waters (Olesiuk et al. 1999, Olesiuk 2010). The surveys still reflect all of the inherent inaccuracy of visual counts for animals whose haul-out behaviour varies with tidal and environmental conditions, uncertainty in correction factors, and immigration and emigration from the census area (Olesiuk 2010). This assessment also represents a break in continuity due to program staff turnover and the integration of key updates at all levels of the process: survey extent and methods, handling of data, adjusting/correcting the counts for abundance estimation, error estimates and modelling trends in abundance. While census flights were completed in the same manner, the number of observers (not counting the pilot) has generally increased over time from one individual to two (at times three), as has the number of individuals undertaking counting of photos. Updated camera equipment and ultra high-resolution screens have been added to capture and count images of oftentimes cryptic individuals. In concert, these refinements have likely resulted in more accurate and precise counts overall. The recast of previous abundance estimates based on updated shoreline densities, and haul-out CFs with new deployments of satellite transmitters and different means of error estimation likely improved the time series. However, outstanding issues remain. Future studies are proposed to evaluate changes in pupping phenology, and to more clearly define the difference in the timing of pupping between northern and southern regions of BC. Further, it is unclear if seal haul-out behaviour, particularly as it relates to survey conditions, is consistent across BC. The development of correction factors for areas outside of the SOG are needed, and are considered a key objective for future assessments.

6. ACKNOWLEDGEMENTS

We are grateful to Dr. Michael Bigg and Peter Olesiuk for establishing the standardized harbour seal surveys used for this assessment; to Peter Olesiuk for his work to continue the time series, expand survey coverage, develop correction factors and complete historic reconstructions of harbour seal populations in BC; and to Graeme Ellis for readily sharing his invaluable knowledge and experience. Thank-you to the host of individuals who assisted with planning and completing survey flights and seal captures; and to Elise Keppel, Kai Meyer, Colin Novak, Johanna Fee, Brianna Cairns and Lisa Spaven who assisted with the counts. We are grateful to the DFO GIS team Cole Fields, Kayleigh Gillespie and Michelle Bigg for graciously taking on our pathfinding project, aiming to meet our timelines. Thanks to DFO Species at Risk Program, World Class Measures (WCM), DFO OPP Coastal Environmental Baseline Program, Gwaii Haanas National Park Reserve, National Marine Conservation Area Reserve, and Haida Heritage Site (Parks Canada) and the Centre of Expertise in Marine Mammalogy (CEMAM) for providing funds to conduct surveys and subsequent analysis. A special thank-you to pilots Louis Rouleau and Katie Waito for their aviation skills, sharp eyes, and continued enthusiasm, while keeping us safe when flying the surveys.

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8. TABLES

Table 1. Aerial surveys conducted by a Cessna 180 floatplane in BC 2015 – 2019. Primary observers included Sheena Majewski (SPM) and Chad Nordstrom (CN). See Figure 1 for Region and Subregion delineations.

Survey Number	Survey Date	Primary Observer	Distance (km)	Region	Subregion
216	05-Jul-2015	SPM	359	NMC	PFMA_06
217	06-Jul-2015	SPM	707	NMC	PFMA_06
218	07-Jul-2015	SPM	670	NMC	PFMA_06
219	08-Jul-2015	SPM	711	NMC	PFMA_06
220	19-Jul-2015	SPM	544	NMC	PFMA_06
221	21-Jul-2015	SPM	641	NMC	PFMA_06
222	22-Jul-2015	SPM	378	NMC	PFMA_09
223	02-Aug-2015	SPM	543	HG	NHG
224	03-Aug-2015	SPM	633	NMC	PFMA_05
225	04-Aug-2015	SPM	504	NMC	PFMA_05
226	18-Aug-2016	SPM	506	WCVI	SWCVI
227	19-Aug-2016	SPM	348	WCVI	SMWCVI
228	21-Aug-2016	SPM	496	WCVI	SMWCVI
229	22-Aug-2016	SPM	482	WCVI	SMWCV / NMWCVI
230	23-Aug-2016	SPM	387	WCVI	BARKLEY
231	24-Aug-2016	SPM	320	WCVI	BARKLEY
232	02-Sep-2016	SPM	939	WCVI	NMWCVI
233	04-Sep-2016	SPM	452	WCVI	NWCVI
234	05-Sep-2016	SPM	561	WCVI	NMWCVI / NWCVI
235	25-Jun-2017	SPM	520	HG	NWCG
236	26-Jun-2017	SPM	719	HG	NECHG
237	27-Jun-2017	SPM	496	HG	SEHG
238	28-Jun-2017	SPM	857	HG	NWCHG / NHG
239	23-Jul-2017	SPM	628	HG	NHG
240	24-Jul-2017	SPM	802	HG	SHG / SWHG
241	25-Jul-2017	SPM	861	HG	SEHG
242	26-Jul-2017	SPM	565	HG	NWCHG
243	06-Aug-2017	SPM	761	DPASS	DISCOVPASS
244	07-Aug-2017	SPM	613	DPASS	DISCOVPASS
245	18-Aug-2017	SPM	600	DPASS	DISCOVPASS
246	20-Aug-2017	SPM	846	DPASS	DISCOVPASS
247	21-Aug-2017	SPM	506	DPASS	DISCOVPASS
248	22-Aug-2017	SPM	389	DPASS	DISCOVPASS
249	23-Aug-2017	SPM	665	QCS	SWQCS / NEQCS

Survey Number	Survey Date	Primary Observer	Distance (km)	Region	Subregion
250	24-Aug-2017	SPM	770	QCS	BROUGHT / NEQCS
251	06-Sep-2017	SPM	993	QCS	NEQCS
252	07-Sep-2017	SPM	559	QCS	BROUGHT
253	08-Sep-2017	SPM	774	DPASS	DISCOVPASS
254	28-May-2018	SPM	474	NMC	PFMA_03
255	29-May-2018	SPM	796	NMC	SKEENAR
256	30-May-2018	SPM	676	NMC	PFMA_03 / SKEENAR
257	31-May-2018	SPM	732	NMC	PFMA_03 / PFMA_04
258	01-Jun-2018	SPM	287	NMC	PFMA_04 / PFMA_05
259	02-Jun-2018	SPM	843	NMC	SKEENAR
260	03-Jun-2018	SPM	685	NMC	PFMA_05
261	17-Jun-2018	SPM	876	CMC	PFMA_05
262	18-Jun-2018	SPM	769	CMC	PFMA_07
263	01-Jul-2018	SPM	789	CMC	PFMA_06 / PFMA_07
264	02-Jul-2018	SPM	856	CMC	PFMA_09
265	03-Jul-2018	SPM	772	CMC	PFMA_07
266	15-Jul-2018	SPM	852	CMC	PFMA_08
267	16-Jul-2018	SPM	1115	CMC	PFMA_09 / PFMA_10
268	18-Jul-2018	SPM	548	CMC	PFMA_09
269	12-Aug-2019	SPM	887	SOG	BBAY / FRASERR / HOWESD
270	13-Aug-2019	SPM	665	SOG	GULFISL
271	14-Aug-2019	SPM	709	SOG	GULFISL
272	15-Aug-2019	SPM	474	SOG	SGULF
273	25-Aug-2019	SPM	485	SOG	BURRINDARM
274	26-Aug-2019	SPM	750	SOG	NEGULF
275	27-Aug-2019	SPM	685	SOG	NEGULF
276	28-Aug-2019	SPM/CN	732	SOG	NWGULF

Table 2. Deployment history for 32 harbour seals instrumented from 2019 – 2021 in the SOG to update haul-out correction factors for aerial surveys. See Figure 4 for specific locations.

Year	Seal ID	Sex	Class	Preg.	Length (cm)	Weight (kg)	Capture Site	Instrument	Date	Duration (days)	Instrument Recovered
2019	Pv19SOG-01	M	Juv	N	118	42.6	Snake Is	SPOT6-293A	2019-04-02	127	N
2019	Pv19SOG-03	M	Adult	N	158	86.2	Entrance Is Rf	SPOT6-293A	2019-05-09	130	Y
2019	Pv19SOG-04	M	Adult	N	168	90.0	Snake Is	SPOT6-293A	2019-05-26	131	Y
2019	Pv19SOG-05	F	Adult	Y	135	66.2	Orlebar Pt Rf	SPOT6-293A	2019-06-03	98	Y
2019	Pv19SOG-07	F	Adult	Y	152	78.2	Snake Is	SPLASH-297A	2019-06-14	122	Y
2019	Pv19SOG-08	M	Adult	NA	148	88.2	Snake Is	SPLASH-297A	2019-06-17	53	N
2019	Pv19SOG-09	F	Juv	N	114	46.8	Snake Is	SPOT6-293A	2019-07-31	32	Y
2019	Pv19SOG-10	M	Juv	N	112	35.8	Snake Is	SPOT6-293A	2019-07-31	21	Y
2019	Pv19SOG-11	F	Juv	N	108	35.0	Brandon Is	SPLASH-297A	2019-08-01	37	Y
2019	Pv19SOG-12	F	Juv	N	128	50.6	Snake Is	SPLASH-297A	2019-08-02	23	Y
2020	Pv20SOG-01	F	Adult	Y	123	63.2	Snake Is	SPLASH-297A	2020-07-01	64	N
2020	Pv20SOG-02	F	Adult	Y	142	77.4	Orlebar Pt Rf	SPLASH-297A	2020-07-02	70	Y
2020	Pv20SOG-03	F	Adult	Y	120	54.8	Snake Is	SPLASH-297A	2020-07-06	73	Y
2020	Pv20SOG-04	M	Adult	NA	137	58.8	Snake Is	SPLASH-297A	2020-07-07	74	Y
2020	Pv20SOG-05	M	Juv	NA	117	43.0	Snake Is	SPLASH-297A	2020-07-07	45	Y
2020	Pv20SOG-06	F	Adult	Y	138	83.6	Brandon Is	SPLASH-297A	2020-07-08	70	Y
2020	Pv20SOG-07	F	Adult	Y	148	89.2	Snake Is	SPLASH-297A	2020-07-14	52	N
2020	Pv20SOG-08	F	Juv	N	118	47.2	Snake Is	SPLASH-297A	2020-07-21	29	Y
2020	Pv20SOG-09	M	Adult	NA	134	69.8	Snake Is	SPLASH-297A	2020-07-22	69	Y
2020	Pv20SOG-10	M	Adult	NA	165	87.6	Brandon Its	SPLASH-297A	2020-07-23	76	Y
2020	Pv20SOG-11	M	Adult	NA	164	80.0	Orlebar Pt Rf	SPLASH-ECD	2020-07-24	24	Y
2020	Pv20SOG-12	F	Juv	N	125	53.0	Orlebar Pt Rf	SPLASH-297A	2020-07-29	48	Y
2020	Pv20SOG-13	F	Adult	Y	132	68.4	Snake Is	SPLASH-297A	2020-07-31	27	Y
2021	Pv21SOG-01	F	Adult	Possibly	137	70.4	Snake Is	SPLASH-297A	2021-05-25	92	Y
2021	Pv21SOG-02	F	Juv	N	128	50.6	Snake Is	SPLASH-297A	2021-06-02	83	Y
2021	Pv21SOG-03	F	Adult	Possibly	148	80.2	Snake Is	SPLASH-297A	2021-06-08	65	Y

Year	Seal ID	Sex	Class	Preg.	Length (cm)	Weight (kg)	Capture Site	Instrument	Date	Duration (days)	Instrument Recovered
2021	Pv21SOG-04	F	Adult	Possibly	134	65.8	Snake Is	SPLASH-297A	2021-06-14	66	N
2021	Pv21SOG-05	F	Adult	Y	139	73.4	Snake Is	SPLASH-297A	2021-06-16	9	Y
2021	Pv21SOG-06	F	Juv	N	123	41.4	Snake Is	SPLASH-297A	2021-06-17	85	Y
2021	Pv21SOG-07	F	Adult	Y	136	70.4	Snake Is	SPLASH-297A	2021-06-23	78	Y
2021	Pv21SOG-08	M	Adult	NA	147	91.2	Orlebar Pt Rf	SPLASH-297A	2021-07-21	89	N
2021	Pv21SOG-09	M	Juv	N	112	38.2	Entrance Is Rf	SPLASH-297A	2021-07-22	24	Y

Table 3. Mean seal density (seal/km) by subregion (see Figure 1b for subregion designations). Year attributed is the survey year with greatest survey coverage.

Region	Attributed Year	Subregion	Density assigned
WCVI	1993	BARKLEY	1.60
WCVI	1994	BARKLEY	1.70
WCVI	1995	BARKLEY	1.80
WCVI	1996	BARKLEY	1.86
QCS	1988	BROUGHT	3.33
SOG	1966	FRASERR	2.00
SOG	1983	FRASERR	4.20
SOG	1966	GULFISL	1.00
SOG	1983	GULFISL	2.94
SOG	1984	GULFISL	3.60
SOG	1985	GULFISL	4.23
SOG	1987	GULFISL	5.97
SOG	1966	HOWESD	0.30
SOG	1976	HOWESD	0.59
SOG	1983	HOWESD	1.32
SOG	2003	HOWESD	3.86
HG	1986	NECHG	1.87
HG	1992	NECHG	1.25
HG	1994	NECHG	2.59
HG	2008	NECHG	1.77
SOG	1966	NEGULF	0.30
SOG	1973	NEGULF	0.50
SOG	1974	NEGULF	0.50
SOG	1982	NEGULF	1.18
SOG	1984	NEGULF	2.30
SOG	1985	NEGULF	2.76
SOG	1986	NEGULF	3.12
QCS	1988	NEQCS	2.78
QCS	1996	NEQCS	3.44
HG	1992	NHG	4.70
HG	1994	NHG	6.51
WCVI	1976	NMWCVI	0.30
WCVI	1987	NMWCVI	2.00
WCVI	1994	NMWCVI	1.98

Region	Attributed Year	Subregion	Density assigned
WCVI	1996	NMWCVI	1.10
HG	1986	NWCHG	1.87
HG	1992	NWCHG	2.00
HG	1994	NWCHG	2.00
WCVI	1976	NWCVI	0.30
WCVI	1987	NWCVI	2.00
WCVI	1993	NWCVI	1.50
WCVI	1994	NWCVI	1.50
WCVI	2007	NWCVI	1.87
SOG	1966	NWGULF	0.30
SOG	1973	NWGULF	0.36
SOG	1982	NWGULF	1.64
SOG	1984	NWGULF	3.36
SOG	1986	NWGULF	5.30
SOG	1987	NWGULF	6.80
SOG	1996	NWGULF	0.36
NMC	1977	PFMA_03	0.77
NMC	1983	PFMA_03	0.77
NMC	1987	PFMA_03	0.77
NMC	1998	PFMA_03	1.75
NMC	1977	PFMA_04	0.77
NMC	1983	PFMA_04	0.77
NMC	1987	PFMA_04	0.77
NMC	1998	PFMA_04	0.77
NMC	1999	PFMA_04	1.75
NMC	1977	PFMA_05	0.77
NMC	1983	PFMA_05	0.77
NMC	1987	PFMA_05	0.77
NMC	1998	PFMA_05	0.77
NMC	1999	PFMA_05	0.77
NMC	1977	PFMA_06	0.77
NMC	1983	PFMA_06	0.77
NMC	1987	PFMA_06	0.77
NMC	1998	PFMA_06	0.77
NMC	1999	PFMA_06	1.27
CMC	2000	PFMA_07	0.92
CMC	2000	PFMA_10	1.67

Region	Attributed Year	Subregion	Density assigned
CMC	2000	PMFA_08	0.89
SOG	1966	SGULF	0.30
SOG	1983	SGULF	6.04
SOG	1984	SGULF	6.55
SOG	1985	SGULF	7.06
SOG	1987	SGULF	10.48
SOG	1992	SGULF	15.78
HG	1986	SHG	1.11
HG	1992	SHG	1.11
NMC	1999	SKEENAR	5.05
WCVI	1976	SMWCVI	0.30
WCVI	1987	SMWCVI	2.00
WCVI	1993	SMWCVI	1.50
WCVI	1995	SMWCVI	1.50
WCVI	1996	SMWCVI	1.50
WCVI	1993	SWCVI	1.50
WCVI	1994	SWCVI	1.50
WCVI	1995	SWCVI	1.50
WCVI	1996	SWCVI	1.50
HG	1986	SWHG	0.86
HG	1992	SWHG	0.92
QCS	1996	SWQCS	2.40
QCS	2004	SWQCS	3.06

Table 4. Initial starting parameters for the theta-logistic for subregions of the SOG where N_{init} *= initial harbour seal abundance, K = carrying capacity,* r_m *= the maximum rate of increase and* θ *= the shape of density dependence.*

Subregion	N_{init}	K	r_m	θ
SGULF	329	7200	0.15	5
BBAY	475	1700	0.1	7
FRASERR	284	2100	0.15	5
HOWESD	103	1250	0.13	8
GULFISL	368	10000	0.125	7.5
NWGULF	100	6000	0.2	7.5
NEGULF	335	15000	0.2	8
SOG	2080	40000	0.13	5

Table 5. Survey coverage estimates using GPS tracks from flights conducted between 2003 – 2019. Region/Subregion length is the total shoreline length (km). Region/Subregion covered is the shoreline length (km) surveyed in that Attributed Year. Coverage type is classified as wide (covered >70% of all available shoreline) or discrete (targeted specific locales).

Region	Region Length (km)	Attributed Year	Region Covered (km)	Proportion Region Covered	Subregion	Subregion Length (km)	Subregion Covered (km)	Proportion Subregion Covered	Coverage Type
SOG	4151	2003	2238	0.54	SGULF	519	394	0.76	Wide
SOG	4151	2003	2238	0.54	BBAY	51	46	0.91	Wide
SOG	4151	2003	2238	0.54	FRASERR	336	155	0.46	Wide
SOG	4151	2003	2238	0.54	HOWESD	358	NA	NA	NA
SOG	4151	2003	2238	0.54	GULFISL	976	898	0.92	Wide
SOG	4151	2003	2238	0.54	NWGULF	356	225	0.63	Wide
SOG	4151	2003	2238	0.54	NEGULF	1554	520	0.33	Wide
SOG	4151	2008	3150	0.76	SGULF	519	379	0.73	Wide
SOG	4151	2008	3150	0.76	BBAY	51	42	0.83	Wide
SOG	4151	2008	3150	0.76	FRASERR	336	163	0.48	Wide
SOG	4151	2008	3150	0.76	HOWESD	358	249	0.69	Wide
SOG	4151	2008	3150	0.76	GULFISL	976	894	0.92	Wide
SOG	4151	2008	3150	0.76	NWGULF	356	334	0.94	Wide
SOG	4151	2008	3150	0.76	NEGULF	1554	1091	0.70	Wide
WCVI	5643	2007	4312	0.76	SWCVI	340	246	0.72	Wide
WCVI	5643	2007	4312	0.76	BARKLEY	934	713	0.76	Wide
WCVI	5643	2007	4312	0.76	SMWCVI	1802	1710	0.95	Wide
WCVI	5643	2007	4312	0.76	NMWCVI	1578	1537	0.97	Wide
QCS	3579	2003	392	0.11	BROUGHT	1479	363	0.25	Discrete
QCS	3579	2004	1038	0.29	NEQCS	1339	421	0.31	Discrete
QCS	3579	2004	1038	0.29	BROUGHT	1479	480	0.32	Discrete
DPASS	2484	2003	1045	0.42	DISCOVPASS	2484	1045	0.42	Discrete

Region	Region Length (km)	Attributed Year	Region Covered (km)	Proportion Region Covered	Subregion	Subregion Length (km)	Subregion Covered (km)	Proportion Subregion Covered	Coverage Type
CMC	7177	2004	1989	0.28	PMFA_08	1633	397	0.24	Discrete
CMC	7177	2004	1989	0.28	PFMA_09	954	697	0.73	Wide
CMC	7177	2004	1989	0.28	PFMA_10	616	546	0.89	Wide
CMC	7177	2005	3572	0.50	PFMA_07	3974	3377	0.85	Wide
CMC	7177	2006	1685	0.23	PMFA_08	1633	1155	0.71	Wide
CMC	7177	2006	1685	0.23	PFMA_09	954	140	0.15	Discrete
NMC	10220	2005	4781	0.47	SKEENAR	682	500	0.73	Wide
NMC	10220	2005	4781	0.47	PFMA_03	1561	209	0.13	Discrete
NMC	10220	2005	4781	0.47	PFMA_04	1083	1006	0.93	Wide
NMC	10220	2005	4781	0.47	PFMA_05	2679	2341	0.87	Wide
NMC	10220	2005	4781	0.47	PFMA_06	4216	724	0.17	Discrete
HG	4580	2008	3505	0.77	SEHG	1629	1423	0.87	Wide
HG	4580	2008	3505	0.77	NEHG	757	464	0.61	Wide
HG	4580	2008	3505	0.77	SHG	395	309	0.78	Wide
HG	4580	2008	3505	0.77	SWHG	463	395	0.85	Wide
HG	4580	2008	3505	0.77	NWHG	1336	913	0.68	Wide
QCS	3579	2009	725	0.20	BROUGHT	1479	689	0.47	Discrete
QCS	3579	2012	829	0.23	NEQCS	1339	722	0.54	Discrete
DPASS	2484	2009	1325	0.53	DISCOVPASS	2484	1325	0.53	Discrete
NMC	10220	2010	2304	0.23	PFMA_06	4216	2173	0.52	Wide
NMC	10220	2012	687	0.07	PFMA_06	4216	687	0.16	Discrete
HG	4580	2010	534	0.12	NEHG	757	80	0.11	Discrete
HG	4580	2010	534	0.12	NWHG	1336	425	0.32	Discrete
SOG	4151	2014	3128	0.75	SGULF	519	381	0.73	Wide
SOG	4151	2014	3128	0.75	BBAY	51	34	0.66	Wide

Region	Region Length (km)	Attributed Year	Region Covered (km)	Proportion Region Covered	Subregion	Subregion Length (km)	Subregion Covered (km)	Proportion Subregion Covered	Coverage Type
SOG	4151	2014	3128	0.75	FRASERR	336	223	0.66	Wide
SOG	4151	2014	3128	0.75	HOWESD	358	267	0.74	Wide
SOG	4151	2014	3128	0.75	GULFISL	976	868	0.89	Wide
SOG	4151	2014	3128	0.75	NWGULF	356	322	0.90	Wide
SOG	4151	2014	3128	0.75	NEGULF	1554	1034	0.67	Wide
SOG	4235	2019	3705	0.87	SGULF	519	387	0.75	Wide
SOG	4235	2019	3705	0.87	BBAY	51	49	0.95	Wide
SOG	4235	2019	3705	0.87	FRASERR	336	242	0.72	Wide
SOG	4235	2019	3705	0.87	HOWESD	358	343	0.96	Wide
SOG	4235	2019	3705	0.87	GULFISL	976	960	0.98	Wide
SOG	4235	2019	3705	0.87	NWGULF	356	334	0.94	Wide
SOG	4235	2019	3705	0.87	NEGULF	1418	1218	0.86	Wide
SOG	4235	2019	3705	0.87	BURRINDARM	220	172	0.78	Wide
WCVI	5643	2016	4508	0.80	SWCVI	340	234	0.69	Wide
WCVI	5643	2016	4508	0.80	BARKLEY	934	699	0.75	Wide
WCVI	5643	2016	4508	0.80	SMWCVI	1802	1422	0.79	Wide
WCVI	5643	2016	4508	0.80	NMWCVI	1578	1351	0.86	Wide
WCVI	5643	2016	4508	0.80	NWCVI	989	803	0.81	Wide
WCVI	5643	2019	77	0.01	SWCVI	340	77	0.23	Discrete
QCS	3579	2017	2902	0.81	SWQCS	761	739	0.97	Wide
QCS	3579	2017	2902	0.81	NEQCS	1339	1044	0.78	Wide
QCS	3579	2017	2902	0.81	BROUGHT	1479	1119	0.76	Wide
DPASS	2620	2017	2472	0.94	DISCOVPASS	2620	2472	0.94	Wide
CMC	7177	2018	6425	0.90	PFMA_07	3974	3486	0.88	Wide
CMC	7177	2018	6425	0.90	PMFA_08	1633	1544	0.95	Wide

Region	Region Length (km)	Attributed Year	Region Covered (km)	Proportion Region Covered	Subregion	Subregion Length (km)	Subregion Covered (km)	Proportion Subregion Covered	Coverage Type
CMC	7177	2018	6425	0.90	PFMA_09	954	845	0.89	Wide
CMC	7177	2018	6425	0.90	PFMA_10	616	549	0.89	Wide
NMC	10220	2015	3660	0.36	PFMA_05	2679	856	0.32	Wide
NMC	10220	2015	3660	0.36	PFMA_06	4216	2724	0.65	Wide
NMC	10220	2018	5318	0.52	SKEENAR	682	641	0.94	Wide
NMC	10220	2018	5318	0.52	PFMA_03	1561	889	0.57	Wide
NMC	10220	2018	5318	0.52	PFMA_04	1083	1063	0.98	Wide
NMC	10220	2018	5318	0.52	PFMA_05	2679	2041	0.76	Wide
HG	4580	2015	479	0.10	NHG	968	346	0.36	Discrete
HG	4580	2017	4288	0.94	SEHG	984	951	0.97	Wide
HG	4580	2017	4288	0.94	NHG	968	882	0.91	Wide
HG	4580	2017	4288	0.94	SHG	395	388	0.98	Wide
HG	4580	2017	4288	0.94	SWHG	463	448	0.97	Wide
HG	4580	2017	4288	0.94	NWCHG	1115	994	0.89	Wide
HG	4580	2017	4288	0.94	NECHG	655	626	0.95	Wide

Table 6. GIS pathfinding estimates for historical surveys without GPS tracks (1966 – 2000) and an additional 38 estimates from 2003 – 2008 for error checking.

Region	Year	Subregion	Subregion Length (km)	Subregion covered (km)	Proportion Subregion covered	GPS Track Available	Coverage Type
SOG	1966	SGULF	520	198	0.38	N	Discrete
SOG	1973	SGULF	520	402	0.77	N	Wide
SOG	1974	SGULF	520	402	0.77	N	Wide
SOG	1976	SGULF	520	218	0.42	N	Wide
SOG	1982	SGULF	520	402	0.77	N	Wide
SOG	1986	SGULF	520	402	0.77	N	Wide
SOG	1988	SGULF	520	402	0.77	N	Wide
SOG	1990	SGULF	520	385	0.74	N	Wide
SOG	1994	SGULF	520	385	0.74	N	Wide
SOG	1996	SGULF	520	385	0.74	N	Wide
SOG	1998	SGULF	520	333	0.64	N	Wide
SOG	2000	SGULF	520	333	0.64	N	Wide
SOG	2003	SGULF	520	400	0.77	Y	Wide
SOG	2008	SGULF	520	344	0.66	Y	Wide
SOG	1966	BBAY	51	40	0.78	N	Wide
SOG	1973	BBAY	51	40	0.78	N	Wide
SOG	1974	BBAY	51	40	0.78	N	Wide
SOG	1976	BBAY	51	40	0.78	N	Wide
SOG	1982	BBAY	51	40	0.78	N	Wide
SOG	1983	BBAY	51	40	0.78	N	Wide
SOG	1984	BBAY	51	40	0.78	N	Wide
SOG	1985	BBAY	51	40	0.78	N	Wide
SOG	1986	BBAY	51	40	0.78	N	Wide
SOG	1987	BBAY	51	40	0.78	N	Wide
SOG	1988	BBAY	51	40	0.78	N	Wide
SOG	1990	BBAY	51	40	0.78	N	Wide
SOG	1992	BBAY	51	40	0.78	N	Wide
SOG	1994	BBAY	51	40	0.78	N	Wide
SOG	1996	BBAY	51	40	0.78	N	Wide
SOG	1998	BBAY	51	40	0.78	N	Wide
SOG	2000	BBAY	51	40	0.78	N	Wide
SOG	2003	BBAY	51	40	0.78	Y	Wide

Region	Year	Subregion	Subregion Length (km)	Subregion covered (km)	Proportion Subregion covered	GPS Track Available	Coverage Type
SOG	2008	BBAY	51	34	0.67	Y	Wide
SOG	1966	FRASERR	332	234	0.7	N	Wide
SOG	1973	FRASERR	332	226	0.68	N	Wide
SOG	1974	FRASERR	332	234	0.7	N	Wide
SOG	1976	FRASERR	332	234	0.7	N	Wide
SOG	1982	FRASERR	332	234	0.7	N	Wide
SOG	1984	FRASERR	332	234	0.7	N	Wide
SOG	1985	FRASERR	332	234	0.7	N	Wide
SOG	1986	FRASERR	332	234	0.7	N	Wide
SOG	1987	FRASERR	332	234	0.7	N	Wide
SOG	1988	FRASERR	332	234	0.7	N	Wide
SOG	1990	FRASERR	332	234	0.7	N	Wide
SOG	1992	FRASERR	332	234	0.7	N	Wide
SOG	1994	FRASERR	332	234	0.7	N	Wide
SOG	1996	FRASERR	332	234	0.7	N	Wide
SOG	1998	FRASERR	332	234	0.7	N	Wide
SOG	2000	FRASERR	332	234	0.7	N	Wide
SOG	2003	FRASERR	332	222	0.67	Y	Wide
SOG	2008	FRASERR	332	234	0.7	Y	Wide
SOG	1973	HOWESD	359	299	0.83	N	Wide
SOG	1974	HOWESD	359	299	0.83	N	Wide
SOG	1982	HOWESD	359	299	0.83	N	Wide
SOG	1984	HOWESD	359	299	0.83	N	Wide
SOG	1985	HOWESD	359	299	0.83	N	Wide
SOG	1986	HOWESD	359	299	0.83	N	Wide
SOG	1987	HOWESD	359	299	0.83	N	Wide
SOG	1988	HOWESD	359	299	0.83	N	Wide
SOG	1990	HOWESD	359	256	0.71	N	Wide
SOG	1992	HOWESD	359	256	0.71	N	Wide
SOG	1994	HOWESD	359	256	0.71	N	Wide
SOG	1996	HOWESD	359	255	0.71	N	Wide
SOG	1998	HOWESD	359	255	0.71	N	Wide
SOG	2000	HOWESD	359	255	0.71	N	Wide
SOG	2008	HOWESD	359	256	0.71	Y	Wide

Region	Year	Subregion	Subregion Length (km)	Subregion covered (km)	Proportion Subregion covered	GPS Track Available	Coverage Type
SOG	1966	GULFISL	978	829	0.85	N	Wide
SOG	1973	GULFISL	978	844	0.86	N	Wide
SOG	1974	GULFISL	978	844	0.86	N	Wide
SOG	1976	GULFISL	978	776	0.79	N	Wide
SOG	1982	GULFISL	978	844	0.86	N	Wide
SOG	1986	GULFISL	978	846	0.87	N	Wide
SOG	1988	GULFISL	978	844	0.86	N	Wide
SOG	1990	GULFISL	978	844	0.86	N	Wide
SOG	1992	GULFISL	978	846	0.87	N	Wide
SOG	1994	GULFISL	978	844	0.86	N	Wide
SOG	1996	GULFISL	978	844	0.86	N	Wide
SOG	1998	GULFISL	978	840	0.86	N	Wide
SOG	2000	GULFISL	978	844	0.86	N	Wide
SOG	2003	GULFISL	978	860	0.88	Y	Wide
SOG	2008	GULFISL	978	845	0.86	Y	Wide
SOG	1974	NWGULF	357	346	0.97	N	Wide
SOG	1976	NWGULF	357	349	0.98	N	Wide
SOG	1983	NWGULF	357	349	0.98	N	Wide
SOG	1985	NWGULF	357	349	0.98	N	Wide
SOG	1988	NWGULF	357	349	0.98	N	Wide
SOG	1990	NWGULF	357	349	0.98	N	Wide
SOG	1992	NWGULF	357	349	0.98	N	Wide
SOG	1994	NWGULF	357	349	0.98	N	Wide
SOG	1996	NWGULF	357	349	0.98	N	Wide
SOG	1998	NWGULF	357	349	0.98	N	Wide
SOG	2000	NWGULF	357	338	0.95	N	Wide
SOG	2003	NWGULF	357	259	0.73	Y	Wide
SOG	2008	NWGULF	357	349	0.98	Y	Wide
SOG	1976	NEGULF	1557	1152	0.74	N	Wide
SOG	1983	NEGULF	1557	1152	0.74	N	Wide
SOG	1987	NEGULF	1557	1138	0.73	N	Wide
SOG	1988	NEGULF	1557	1134	0.73	N	Wide
SOG	1990	NEGULF	1557	1136	0.73	N	Wide
SOG	1992	NEGULF	1557	1137	0.73	N	Wide

Region	Year	Subregion	Subregion Length (km)	Subregion covered (km)	Proportion Subregion covered	GPS Track Available	Coverage Type
SOG	1994	NEGULF	1557	1138	0.73	N	Wide
SOG	1996	NEGULF	1557	1131	0.73	N	Wide
SOG	1998	NEGULF	1557	1093	0.7	N	Wide
SOG	2000	NEGULF	1557	971	0.62	N	Wide
SOG	2003	NEGULF	1557	1094	0.7	Y	Wide
SOG	2008	NEGULF	1557	1169	0.75	Y	Wide
WCVI	1976	SWCVI	340	269	0.79	N	Wide
WCVI	1987	SWCVI	340	269	0.79	N	Wide
WCVI	2007	SWCVI	340	269	0.79	Y	Wide
WCVI	1976	BARKLEY	936	535	0.57	N	Wide
WCVI	1987	BARKLEY	936	535	0.57	N	Wide
WCVI	2007	BARKLEY	936	535	0.57	Y	Wide
WCVI	1994	SMWCVI	1805	495	0.27	N	Discrete
WCVI	2007	SMWCVI	1805	941	0.52	Y	Wide
WCVI	1993	NMWCVI	1581	631	0.4	N	Wide
WCVI	1995	NMWCVI	1581	759	0.48	N	Wide
WCVI	2007	NMWCVI	1581	841	0.53	Y	Wide
WCVI	1995	NWCVI	991	307	0.31	N	Discrete
WCVI	1996	NWCVI	991	382	0.39	N	Discrete
QCS	1988	SWQCS	763	428	0.56	N	Discrete
QCS	1989	SWQCS	763	551	0.72	N	Wide
QCS	1989	NEQCS	1341	207	0.15	N	Discrete
QCS	2004	NEQCS	1341	335	0.25	Y	Discrete
QCS	1989	BROUGHT	1481	238	0.16	N	Discrete
QCS	1996	BROUGHT	1481	416	0.28	N	Discrete
QCS	2003	BROUGHT	1481	374	0.25	Y	Discrete
QCS	2004	BROUGHT	1481	290	0.2	Y	Discrete
DPASS	2003	DISCOVPASS	2488	731	0.29	Y	Discrete
CMC	2005	PFMA_07	3961	2381	0.6	Y	Wide
CMC	2004	PMFA_08	1636	259	0.16	Y	Discrete
CMC	2006	PMFA_08	1636	1145	0.7	Y	Wide
CMC	2000	PFMA_09	956	592	0.62	N	Wide
CMC	2004	PFMA_09	956	545	0.57	Y	Wide
CMC	2006	PFMA_09	956	46	0.05	Y	Discrete

Region	Year	Subregion	Subregion Length (km)	Subregion covered (km)	Proportion Subregion covered	GPS Track Available	Coverage Type
CMC	2004	PFMA_10	617	452	0.73	Y	Wide
NMC	1977	SKEENAR	683	478	0.7	N	Wide
NMC	1983	SKEENAR	683	478	0.7	N	Wide
NMC	1987	SKEENAR	683	478	0.7	N	Wide
NMC	1998	SKEENAR	683	490	0.72	N	Wide
NMC	2005	SKEENAR	683	472	0.69	Y	Wide
NMC	1999	PFMA_03	1565	480	0.31	N	Discrete
NMC	2005	PFMA_03	1565	156	0.1	Y	Discrete
NMC	2005	PFMA_04	1073	825	0.77	Y	Wide
NMC	2005	PFMA_05	2666	1945	0.73	Y	Wide
NMC	2005	PFMA_06	4225	1003	0.24	Y	Discrete
HG	1986	SEHG	986	769	0.78	N	Wide
HG	1992	SEHG	986	695	0.7	N	Wide
HG	1994	SEHG	986	399	0.4	N	Discrete
HG	2008	SEHG	986	769	0.78	Y	Wide
HG	1986	NEHG	759	640	0.84	N	Wide
HG	2008	NEHG	759	599	0.79	Y	Wide
HG	1994	SHG	395	369	0.93	N	Wide
HG	2008	SHG	395	315	0.8	Y	Wide
HG	1994	SWHG	464	405	0.87	N	Wide
HG	2008	SWHG	464	386	0.83	Y	Wide
HG	2008	NWHG	1339	932	0.7	Y	Wide
HG	1986	NECHG	656	487	0.74	N	Wide
HG	1992	NECHG	656	487	0.74	N	Wide
HG	2008	NECHG	656	503	0.77	Y	Wide
QCS	2012	NEQCS	1341	239	0.18	Y	Discrete
QCS	2009	BROUGHT	1481	200	0.14	Y	Discrete
DPASS	2009	DISCOVPASS	2488	702	0.28	Y	Discrete
NMC	2010	PFMA_06	4225	1946	0.46	Y	Wide
NMC	2012	PFMA_06	4225	228	0.05	Y	Discrete
HG	2010	NEHG	759	39	0.05	Y	Discrete
HG	2010	NWHG	1339	88	0.07	Y	Discrete

Table 7. Harbour seal counts and abundance estimates (95% confidence interval, CI) by year and region once corrected for animals at-sea (N, based on a 1 / 0.785 correction factor) and adding seals from un-surveyed area (N-corrected). Model estimates are also provided projecting abundance to 2019 (Modeled N). Density is based on the 2019 projected abundance.

Region	Attributed Year	count	N	Un-surveyed (km)	Added seals	N-corrected	95% CI	Modeled N	95%CI	Density (seal/km)
SOG	2019	26941	35080	30	398	35478	32860-38304	37309	30641-45428	10.5
WCVI	2016	7785	10012	1047	2180	12192	11296-13159	12049	7755-18722	2.1
QCS	2017	3506	4496	587	578	5074	4589-5610	4346	3561-5303	1.2
DP	2017	2438	3228	42	52	3280	2779-3872	4104	1287-13088	1.6
CMC	2018	3018	3963	187	75	4038	3610-4516	4121	2440-6959	0.6
NMC	2018	5998	8420	1499	1569	9989	9199-10846	10677	6875-16580	1.0
HG	2017	9437	14214	127	232	14446	13137-15886	13409	10701-16801	2.9
BC	-	59123	79413	3519	5084	84497	81160-87970	86015	74745-98985	2.3

Table 8. The amount of total data lost and number of unique haul-out events according to the minimum number of seals concurrently providing data, with impacts on the mean proportion of individuals hauled-out (P), variance (var) of P, standard deviation (SD) of P, standard error (SE) of P, and 95% confidence interval (CI) of P, and mean correction factor (CF), when considering any contextual conditions (unfiltered) or when restricting data to survey-like conditions (filtered).

Dataset	Min. Concurr. seals	Cumulative Proportional Data Lost	Unique events	Mean P	Var P	SD P	SE P	95% CI P	Mean CF
Unfiltered	2	0.00	7295	0.317	0.102	0.319	0.004	0.310-0.325	3.151
Unfiltered	3	0.22	5693	0.325	0.094	0.307	0.004	0.317-0.333	3.072
Unfiltered	4	0.32	4981	0.335	0.092	0.303	0.004	0.327-0.344	2.984
Unfiltered (all hours)	5	0.46	3974	0.345	0.088	0.297	0.005	0.336-0.354	2.899
Unfiltered (all hours)	6	0.64	2639	0.345	0.082	0.287	0.006	0.334-0.356	2.901
Unfiltered (all hours)	7	0.74	1891	0.348	0.082	0.287	0.007	0.335-0.361	2.871
Filtered	2	0.00	302	0.761	0.052	0.228	0.013	0.735-0.787	1.314
Filtered	3	0.22	236	0.785	0.035	0.187	0.012	0.761-0.809	1.274
Filtered	4	0.29	214	0.785	0.033	0.181	0.012	0.761-0.809	1.274
Filtered (survey-like)	5	0.46	164	0.791	0.026	0.162	0.013	0.766-0.816	1.264
Filtered (survey-like)	6	0.64	110	0.801	0.022	0.150	0.014	0.773-0.829	1.248
Filtered (survey-like)	7	0.74	79	0.788	0.021	0.144	0.016	0.756-0.821	1.268

Table 9. Regional estimate of abundance (N), density corrections (additional seals) for shoreline missed (km), and the corrected abundance (N_{cor}) for 1966 – 2019. The proportion of the missed regional shoreline, and subsequent contribution of the added seals (%) to N_{cor} is denoted for each year.

Region	Shoreline length (km)	Attributed Year	seal count	N	missed shoreline (km)	Proportion missed (%)	density correction (seals)	N_{cor}	95% CI	Contribution added seals (%)
SOG	**3,563**	1966	651	1,186	2,262	63	894	2,080	1,861-2,324	43
		1973	1,290	2,613	1,752	49	957	3,570	3,181-4,006	27
		1974	1,984	3,315	1,398	39	866	4,181	3,714-4,706	21
		1976	1,586	2,627	796	22	841	3,468	3,099-3,879	24
		1982	3,966	6,606	1,744	49	2,629	9,235	8,287-10,292	28
		1983	2,770	4,595	2,133	60	7,421	12,016	10,803-13,365	62
		1984	1,895	3,216	2,990	84	10,527	13,743	12,361-15,280	77
		1985	2,728	4,583	2,641	74	10,713	15,296	13,735-17,034	70
		1986	5,919	9,829	1,742	49	6,605	16,434	14,712-18,357	40
		1987	4,699	7,848	1,949	55	13,555	21,403	19,196-23,864	63
		1988	14,161	23,423	357	10	2,310	25,733	22,989-28,804	9
		1990	16,744	28,572	414	12	3,296	31,868	28,527-35,600	10
		1992	13,718	22,447	797	22	9,277	31,724	28,286-35,581	29
		1994	23,237	39,207	413	12	4,569	43,776	39,013-49,120	10
		1996	22,663	39,533	417	12	4,178	43,711	38,974-49,024	10

Region	Shoreline length (km)	Attributed Year	seal count	N	missed shoreline (km)	Proportion missed (%)	density correction (seals)	N_{cor}	95% CI	Contribution added seals (%)
		1998	19,541	33,325	493	14	5,019	38,344	34,128-43,081	13
		2000	24,168	39,469	527	15	6,158	45,627	40,398-51,533	13
		2003	23,122	37,783	1,325	37	20,389	58,172	51,220-66,066	35
		2008	21,753	35,935	412	12	3,689	39,624	35,058-44,783	9
		2014	23,561	39,853	435	12	4,976	44,829	39,650-50,685	11
		2019	26,941	35,080	30	1	398	35,478	32,860-38,304	1
WCVI	**5,643**	1976	110	182	4,839	86	1,385	1,567	1,370- 1,793	88
		1987	902	1,483	4,839	86	9,491	10,974	9,691-12,426	86
		1993	754	1,252	5,012	89	8,069	9,321	8,217-10,573	87
		1994	516	951	5,148	91	9,215	10,166	8,946-11,552	91
		1995	437	733	4,577	81	6,319	7,052	6,177-8,051	90
		1996	312	545	5,261	93	7,553	8,098	7,161-9,158	93
		2007	4,706	8,020	1,331	24	1,256	9,276	8,173-10,528	14
		2016	7,785	10,012	1,047	19	2,180	12,192	11,296-13,159	18
QCS	**3,579**	1988	459	751	3,151	88	9,230	9,981	8,518-11,695	92
		1989	1,222	2,323	2,582	72	7,642	9,965	8,503-11,679	77

Region	Shoreline length (km)	Attributed Year	seal count	N	missed shoreline (km)	Proportion missed (%)	density correction (seals)	N_{cor}	95% CI	Contribution added seals (%)
		1996	598	1,050	3,162	88	9,111	10,161	8,720-11,841	90
		2004	2,041	3,917	2,149	60	5,416	9,333	7,798-11,170	58
		2017	3,506	4,496	587	16	578	5,074	4,589-5,610	11
DP	**2,620**	2003	2,038	3,357	1,575	60	5,060	8,417	6,560-10,799	60
		2017	2,438	3,228	42	2	52	3,280	2,779-3,872	2
CMC	**7,177**	2000	207	337	6,585	92	6,345	6,682	5,726-7,797	95
		2005	4,919	8,364	188	3	237	8,601	7,417-9,974	3
		2018	3,018	3,963	187	3	75	4,038	3,610-4,516	2
NMC	**10,220**	1977	351	626	9,734	95	11,152	11,778	10,210-13,587	95
		1983	636	1,142	9,734	95	5,767	6,909	5,991-7,969	83
		1987	1,120	2,004	9,734	95	6,134	8,138	7,067-9,370	75
		1998	1,285	2,104	9,722	95	7,633	9,737	8,509-11,141	78
		1999	535	874	9,740	95	14,722	15,596	13,821-17,599	94
		2005	4,283	7,855	5,439	53	8,907	16,762	14,824-18,954	53
		2018	5,999	8,420	1,499	15	1,569	9,989	9,199-10,846	16
HG	**4,580**	1986	1,961	3,665	2,956	65	5,105	8,770	7,771-9,896	58

Region	Shoreline length (km)	Attributed Year	seal count	N	missed shoreline (km)	Proportion missed (%)	density correction (seals)	N_{cor}	95% CI	Contribution added seals (%)
		1992	1,451	2,904	3,596	79	8,463	11,367	10,007-12,911	74
		1994	1,260	2,469	3,524	77	13,938	16,407	14,390-18,708	85
		2008	6,861	13,582	1,514	33	6,827	20,409	17,735-23,486	33
		2017	9,437	14,214	127	3	232	14,446	13,137-15,886	2

Table 10. Parameter estimates and respective standard errors (se) for theta-logistic models of harbour seal abundance in various subregions of the Strait of Georgia (SOG) and SOG as a whole. See Figure 1 for the delineation of the SOG and subregion codes, and Figure 15 for model predictions. N_{init} = initial population size in 1965, K = carrying capacity, r_m = maximum rate of increase, θ = shape parameter of density dependence.

Subregion	N_{init}	se	K	se	r_m	se	θ	se
SGULF	321	77	7123	0.2	0.1337	0.0219	7.19	26.3
BBAY	482	2	1707	2.8	0.0790	0.0042	32.27	16.7
FRASERR	284	53	2100	0.5	0.0756	0.0134	12.99	34.4
HOWESD	102	21	1249	0.1	0.1760	0.0273	8.49	15.8
GULFISL	363	82	10000	0.1	0.1402	0.0172	6.65	14.3
NWGULF	100	6	6000	0.1	0.2771	0.0047	8.37	1.3
NEGULF	332	0	14999	0.0	0.2234	0.0093	7.17	9.5
SOG	2079	1	40000	0.2	0.1110	0.0145	14.82	61.5

Table 11. The estimated minimum population size (N_{min}) in the year surveyed, coefficient of variation (CV), trend in abundance (see Figure 14), recovery factor (F_r) and PBR estimated for the seven survey regions. Regions are: Strait of Georgia (SOG), west coast of Vancouver Island (WCVI), Queen Charlotte Strait (QCS), Discovery Passage (DP), central mainland coast (CMC), northern mainland coast (NMC), Haida Gwaii (HG) (see Figure 1).

Region	N_{min}	CV	trend	F_r	PBR
SOG	34477	0.034	abundant, stable	1	2069
WCVI	11848	0.034	abundant, stable/increasing	1	711
QCS	4885	0.045	declining	0.5	147
DP	3080	0.075	uncertain	0.1	18
CMC	3871	0.050	uncertainty around abundance, stable/declining	0.25	58
NMC	9683	0.037	abundant, stable/increasing	1	581
HG	13933	0.043	abundant, declining	0.5	418

9. FIGURES

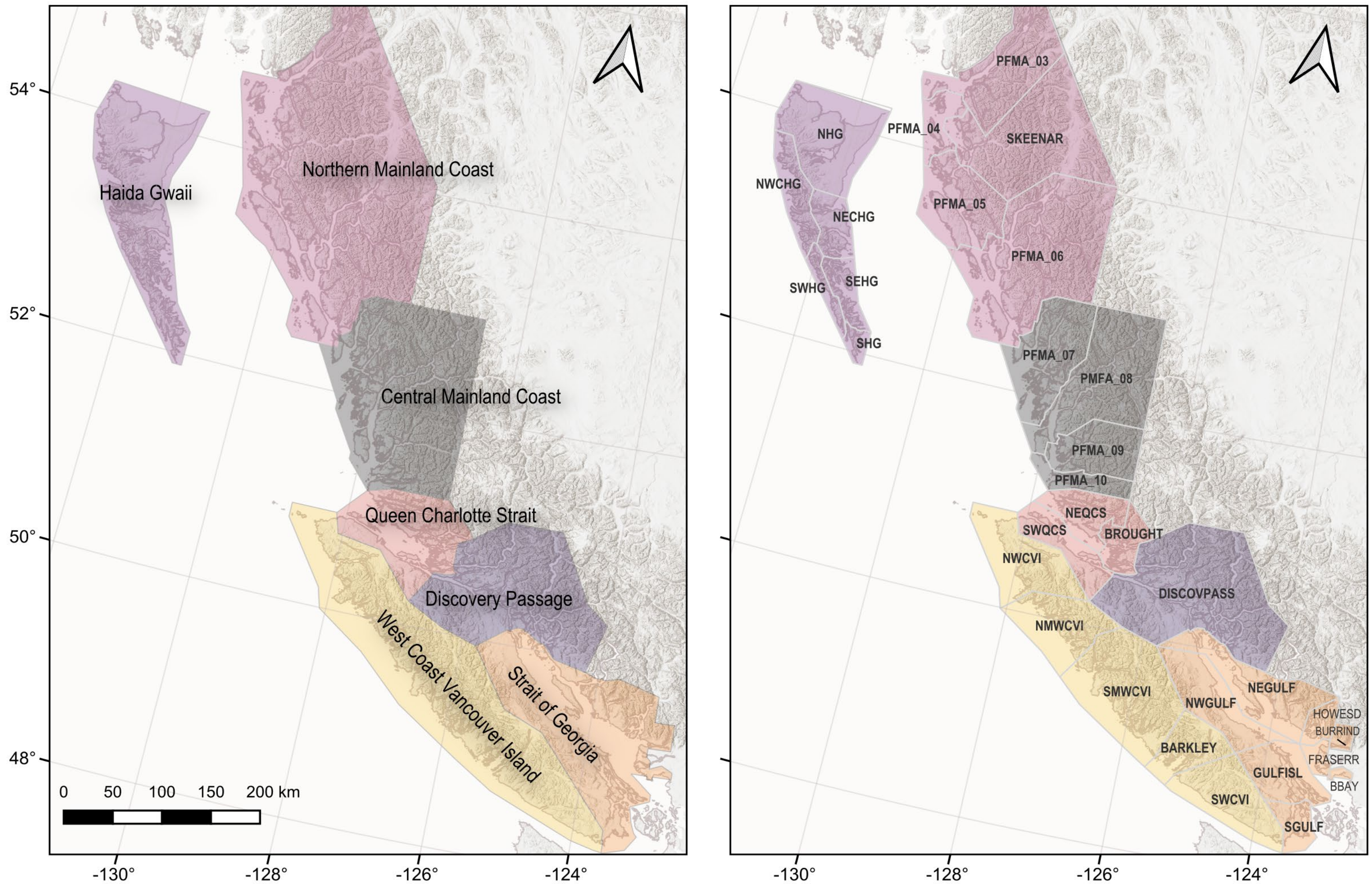

Figure 1. Map of British Columbia coast divided into regions (left) and subregions (right) for harbour seal stock abundance estimation and trends.

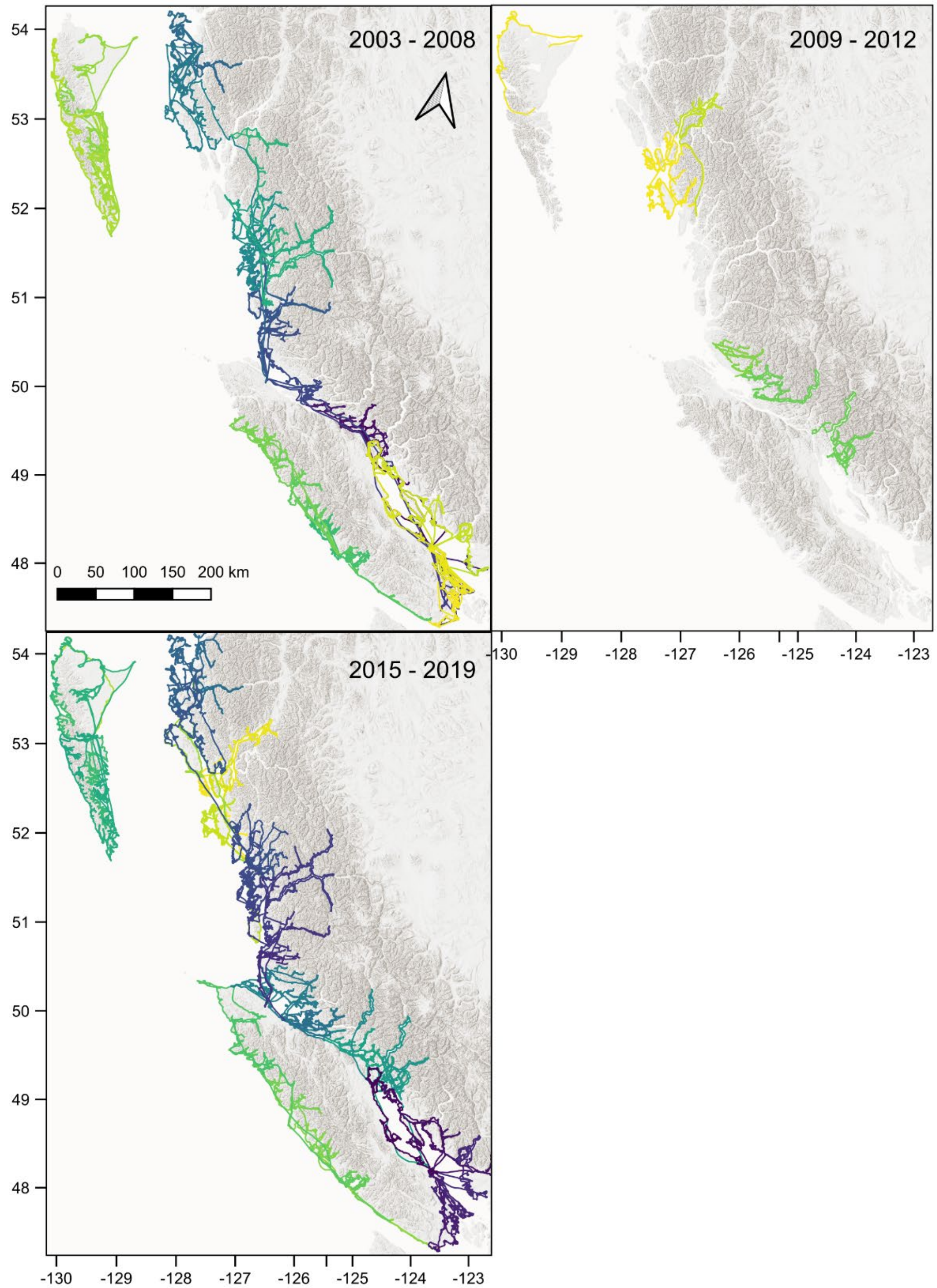

Figure 2. Track lines from harbour seal aerial surveys in BC coastal waters over three time periods: 2003 – 2008, 2009 – 2012, and 2015 – 2019. Lines are coloured by year and subregion for each respective panel. Diverging colours within a region highlight surveys more spaced in time while similar colour bands within a region indicate more synoptic surveys.

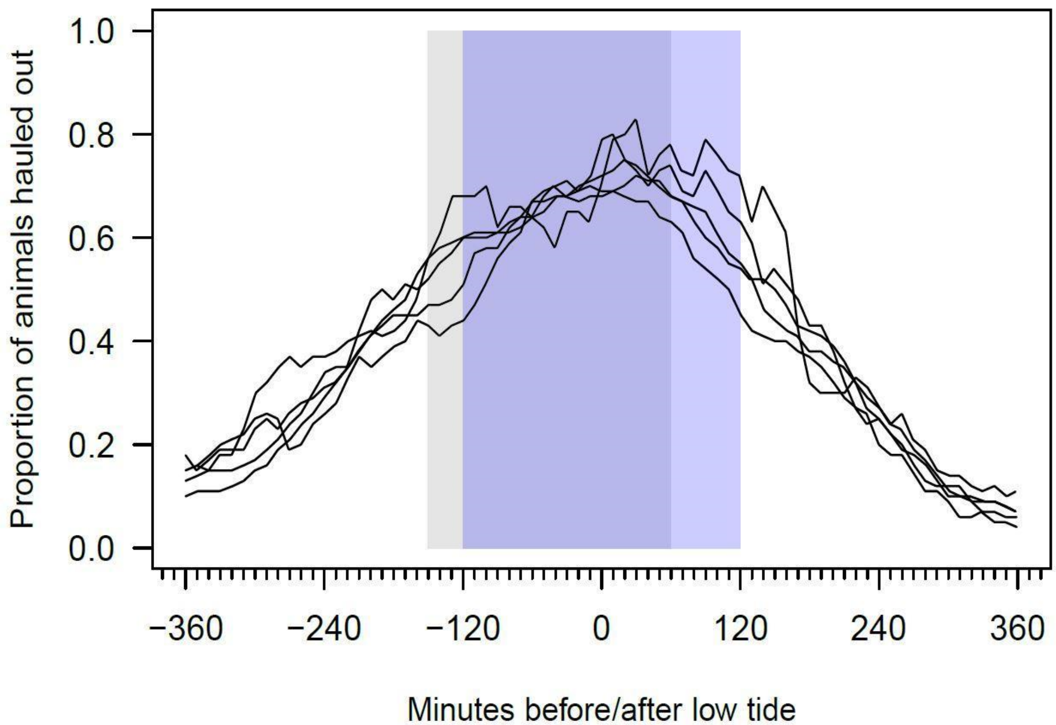

Figure 3. Proportion of animals hauled is shown in relation to time before and after low tide. The grey box shows the survey window of 2.5 hours before to 1 hour after low tide used in surveys prior to 2014. The purple box shows a survey window of 2 hours after low tide, which was used in 2014. The figure is adapted from haul-out response curves adjusted for diel cycle and tidal range presented in Olesiuk (2010).

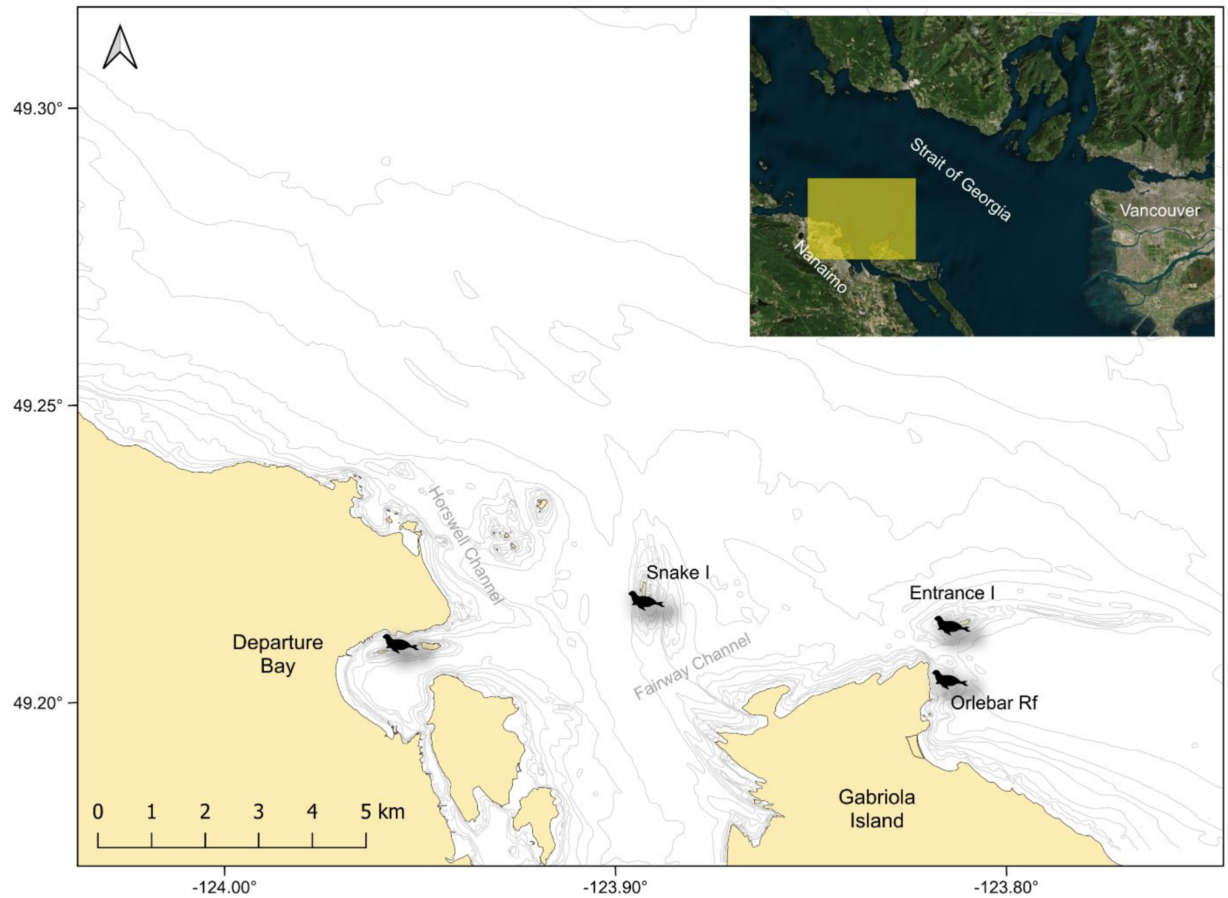

Figure 4. Harbour seal capture sites for satellite tagging in the Strait of Georgia. Researchers based out of Departure Bay.

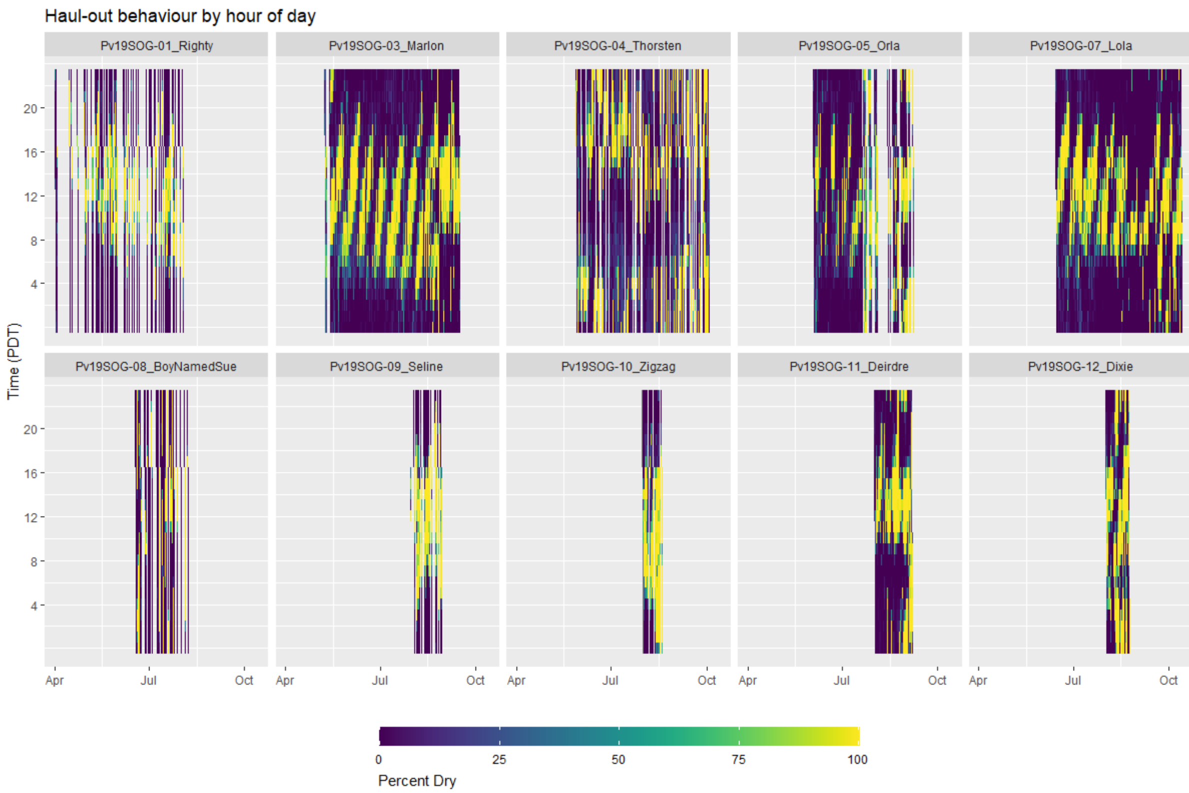

Figure 5. Example percent-dry hourly timelines by day hour (Pacific Daylight Time, PDT) for 10 seals tagged in 2019. Yellow time blocks indicate 100% dry (hauled-out) while indigo blocks indicate 0% dry (at-sea). Cyclical patterns are visible for most seals, showing haul-out behaviour moving progressively later over 10 – 14 days and then re-starting earlier in the day (consistent with the local tidal cycle). Note the off-set cycle observed for seal "Pv19SOG-04" (top-row, middle panel) showing primarily at-sea behaviour when other seals are hauled-out.

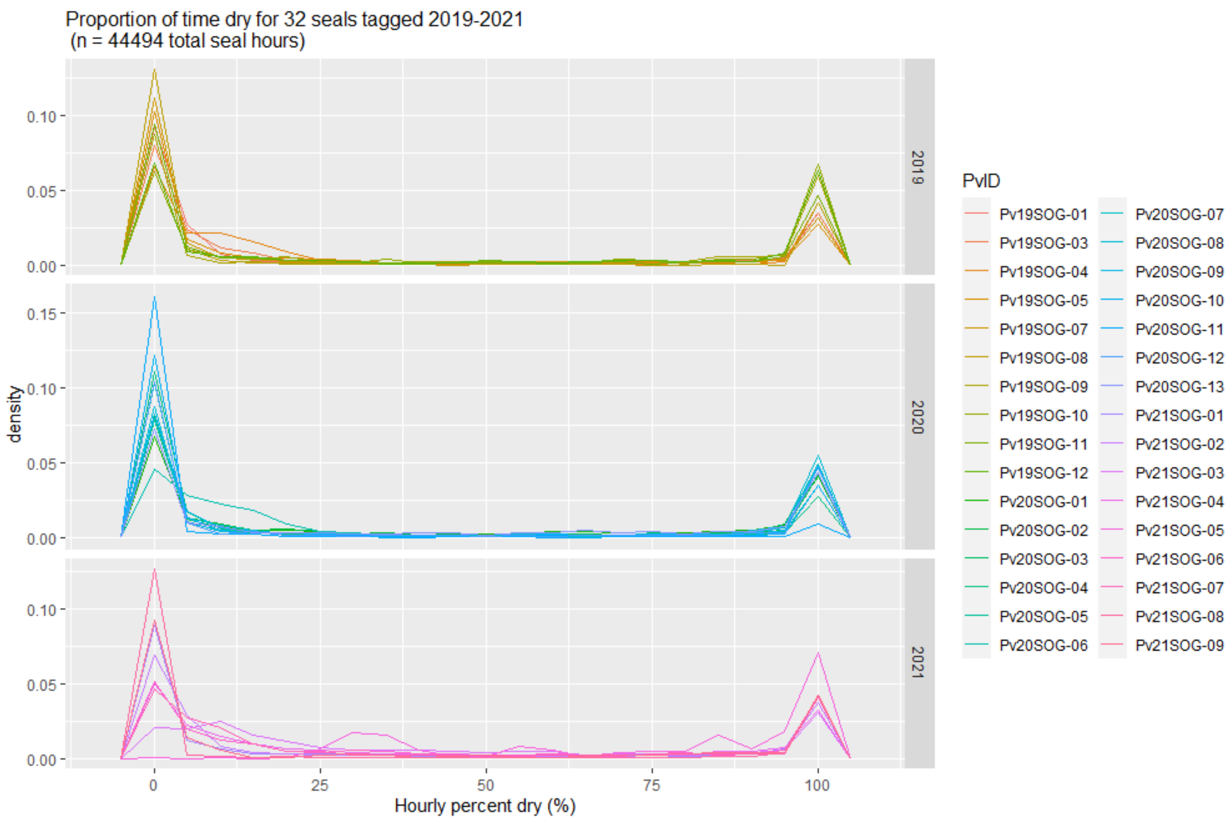

Figure 6. Density plot showing bi-modal distribution of hourly percent dry data (%) for 32 seals tagged during spring and summer of 2019 – 2021. Each coloured line represents one seal. Zero indicates completely wet and 100 represents completely dry for the recorded hour.

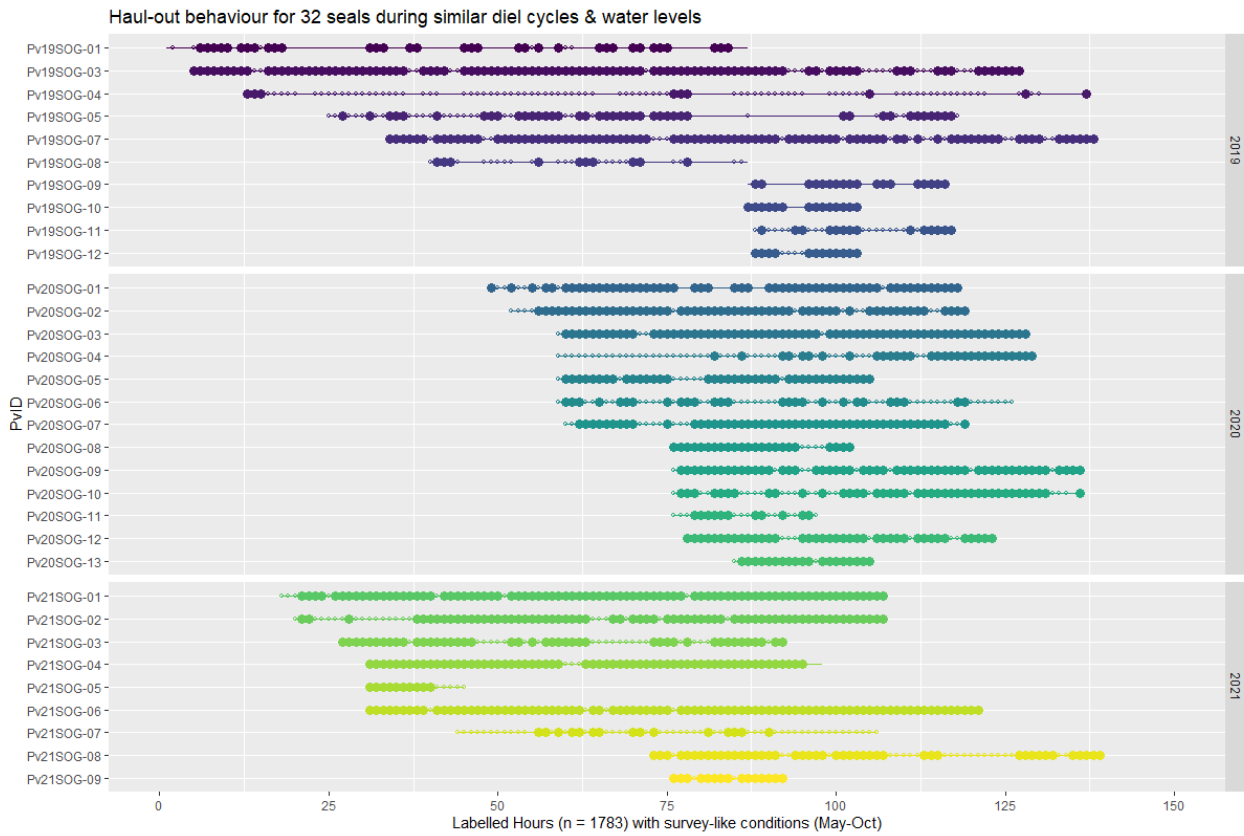

Figure 7. Haul-out behaviour aligned in time for 32 seals tagged in spring and summer from 2019 – 2021. Each coloured line represents one seal. Larger filled circles indicate 'hauled-out' while small open circles indicate 'at-sea' for the given hour. Gaps in a timeline (portion of a line without a circle) indicate 'no-data' for the hour. Note: the breaks on the X-axis are the subset of hours filtered to survey-like conditions (0 – 1.65 m & 08:00 – 12:00 PDT) for the years tagged. Hours are labelled sequentially from 1 to 150 in the order of their occurrence each year to align them across years independent of date. Lines were trimmed for hours prior to tagging and after tags were shed for each seal for visual simplicity.

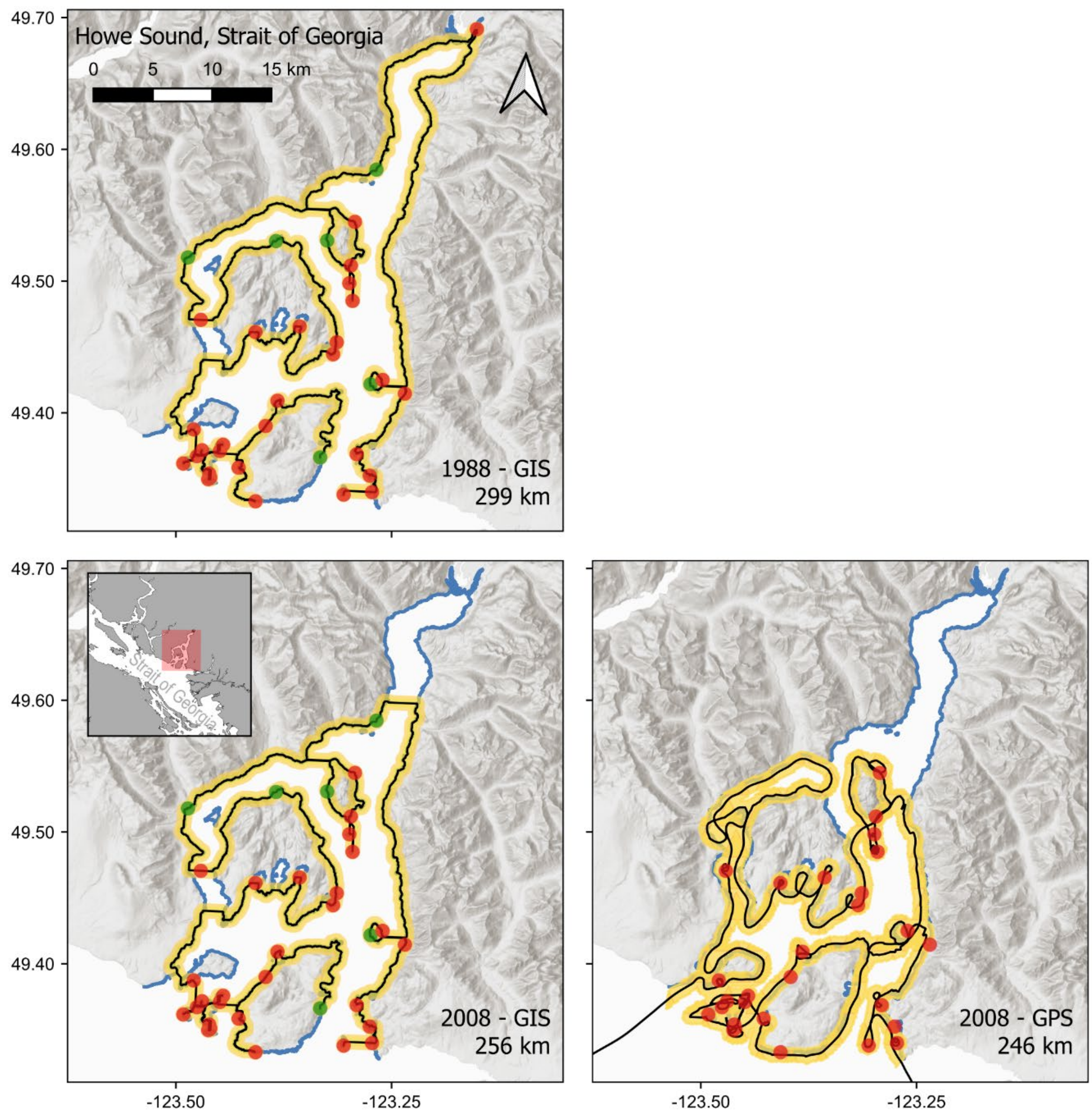

Figure 8. Estimates of shoreline covered during surveys of Howe Sound (Subregion 14) using GIS pathfinding to reconstruct paths from haul-out inventories (left panels) or using track lines generated from GPS receivers carried aboard the survey aircraft (right panel). Buffered estimated paths (black & yellow lines), seal haul-out sites surveyed (red circles), and manually placed anchors for GIS reconstructions (green circles) are displayed. Top left panel: GIS path reconstruction for historical survey without GPS track in 1988 (299 km shoreline covered); lower left panel: GIS path reconstruction for survey in 2008 (256 km); lower right panel: GPS derived path in 2008 (246 km).

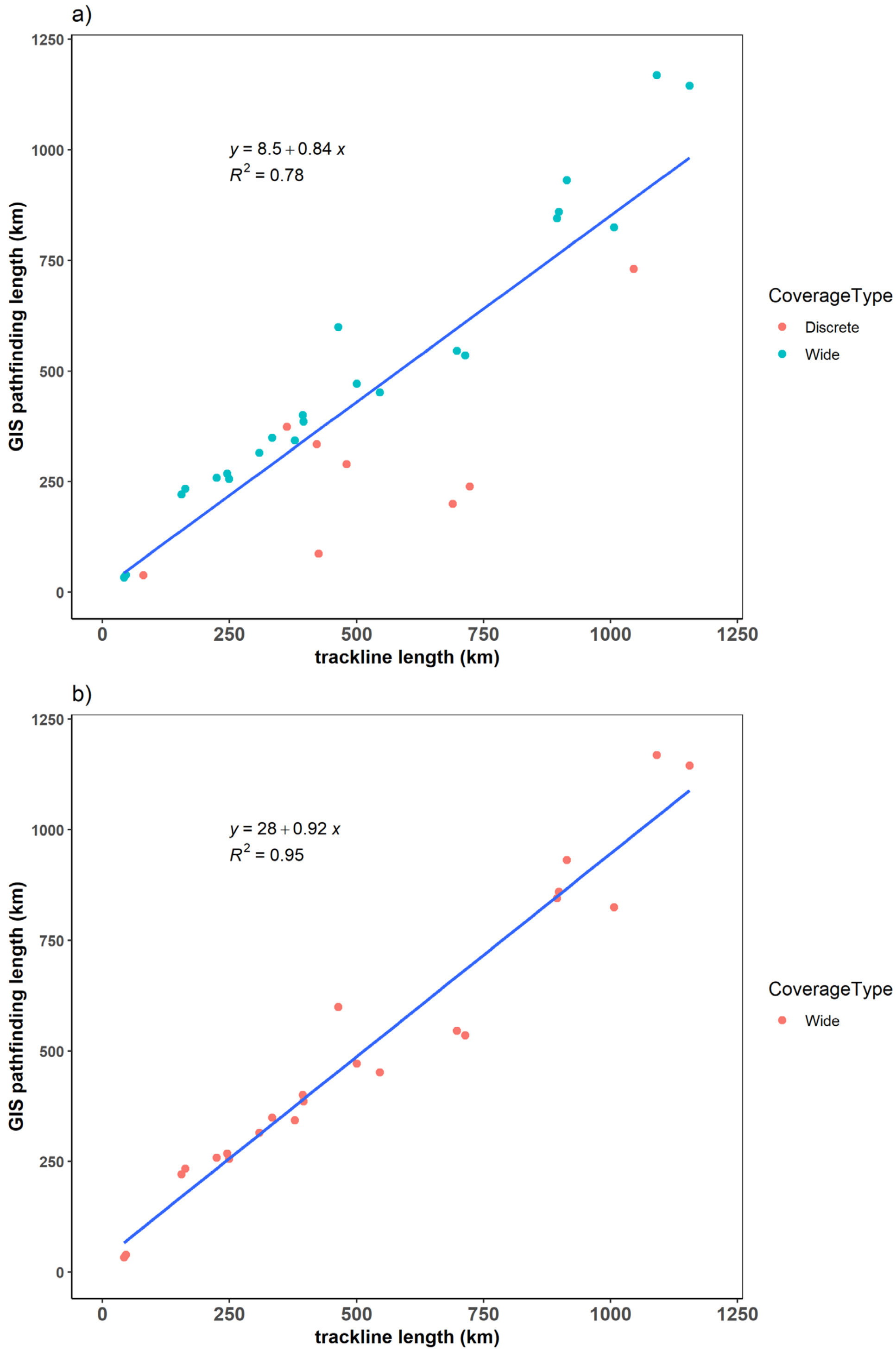

Figure 9. Contrasting survey track line and pathfinding estimates of subregion shoreline length surveyed (km).

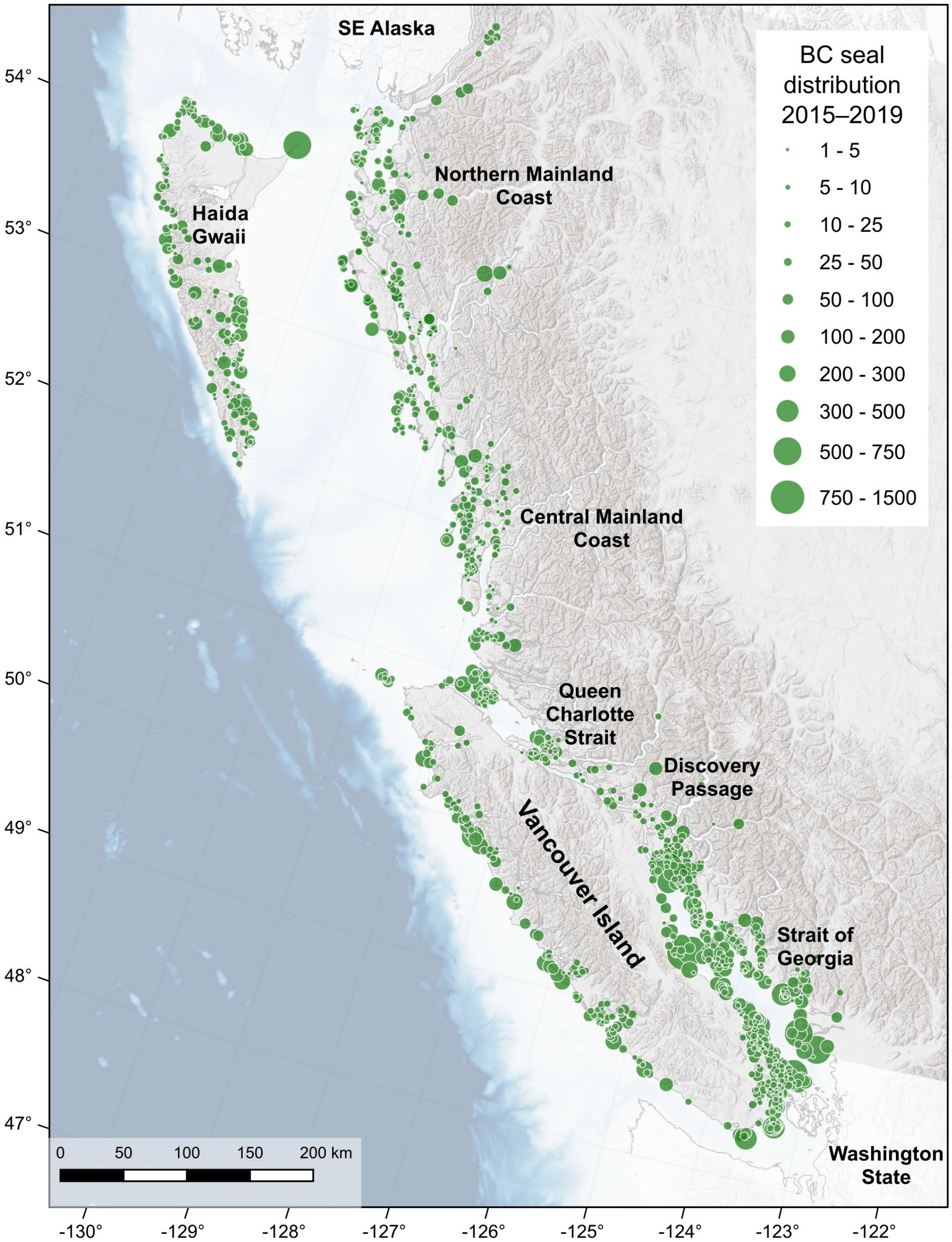

Figure 10. Distribution of harbour seal haul-out sites in BC 2015 – 2019. Circles are scaled relative to the number of animals counted at a site.

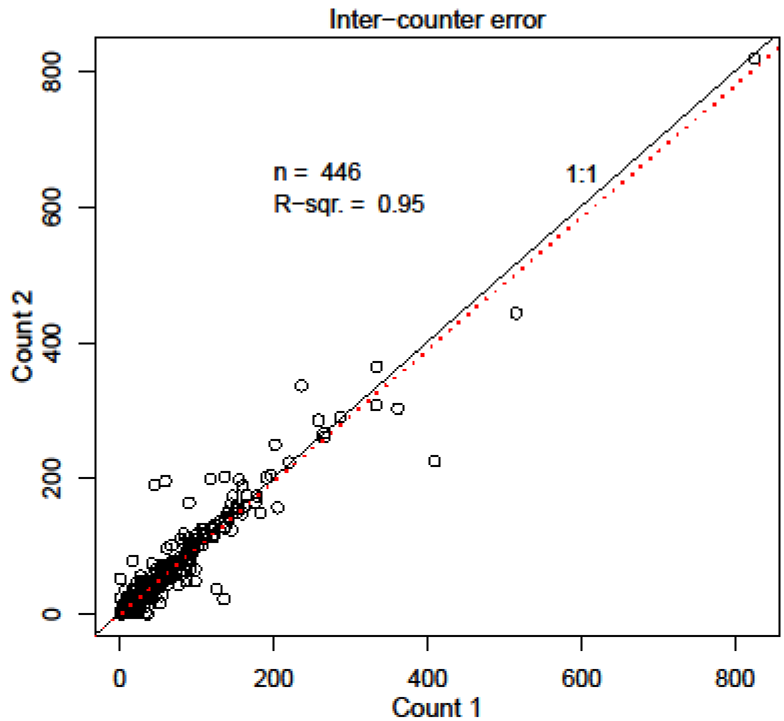

Figure 11. Inter-counter error. The solid line is the 1:1 line and the dotted line is a least squares fit to the data. The fit was not significantly different from the 1:1 line.

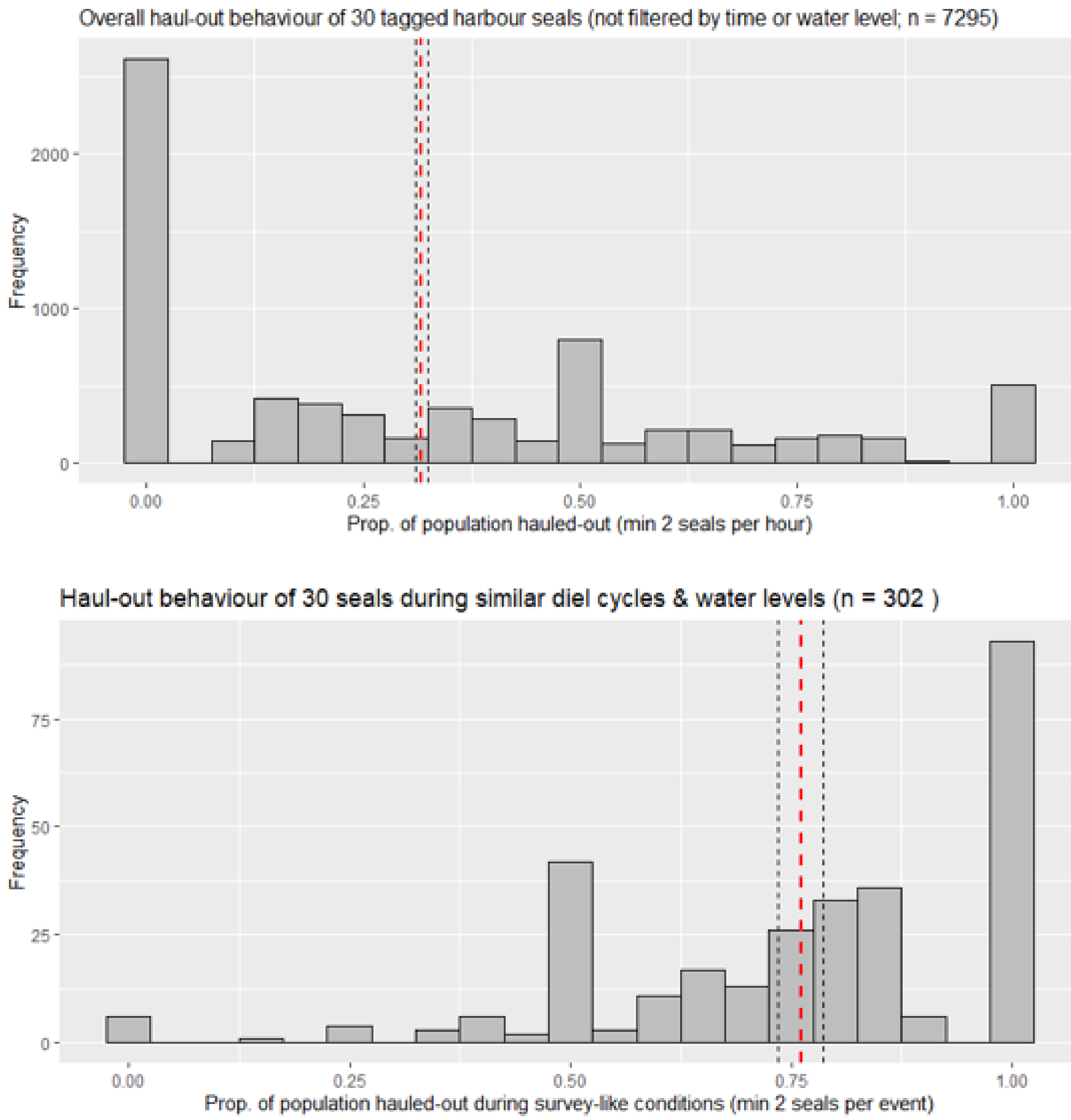

Figure 12. (2 panels) Distribution of the proportion of seals hauled-out at any given period for the full dataset (top panel; not filtered by time or water level) and with the dataset filtered to survey-like conditions (bottom panel; hours between 08:00 and 12:00 PDT & water levels between 0 -1.65 m). Overall means (dashed red lines) and 95% confidence intervals (dashed black lines) overlaid.

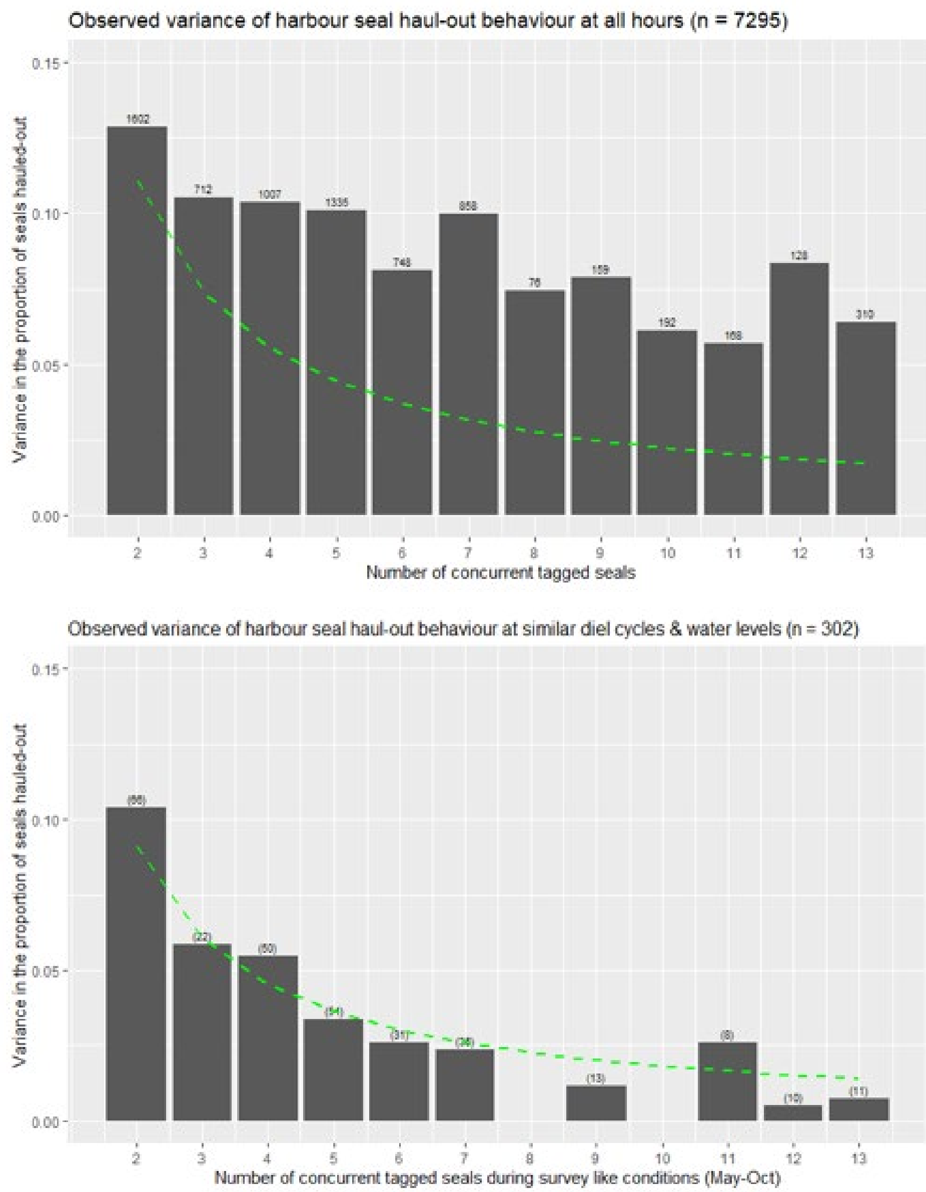

Figure 13. (2 panels) Variance in the proportion of seals hauled-out for varying numbers of concurrent seals (2-13) per event. Top panel: full dataset (not filtered by time or water level); Bottom panel: reduced dataset (filtered to low am waters between 08:15 and 11:45 PDT & water levels between 0 -1.55 m). Dashed green lines are the expected binomial variances calculated using a global mean from pooled data from each dataset. Values above bars are the number of hours (top panel) and number of events (bottom panel) for each collection of concurrent tagged seals which serve as weights when calculating rho (see Appendix B).

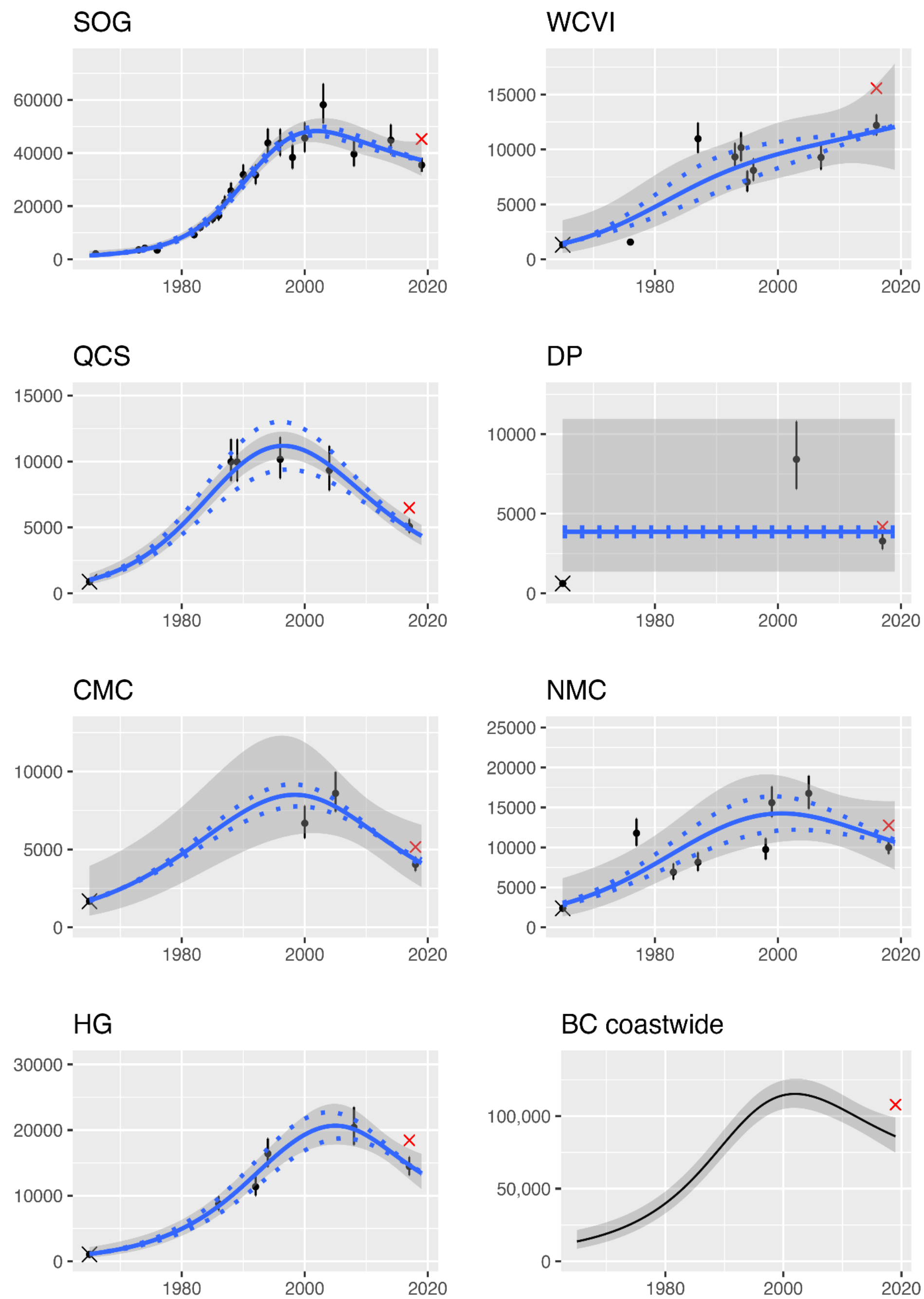

Figure 14. Abundance estimates (95% CI) and trends for seven regions within BC and the BC coastwide. The black "x" in 1965 is an rough estimate of the regional abundance of seal from Olesiuk's (2010) historical reconstruction (see Methods section 2.6). The red "x" indicates the abundance estimate obtained using the 1 / 0.615 correction factor used in previous assessments. Trends in abundance for each region were fit using a generalized additive model (GAM), except for DP where a linear intercept-only model (GLM) was fit. Shaded areas indicate 95% CI to model fit. Dashed lines are trends fitted to abundance estimates where the assumed density of seals in un-surveyed areas was 20% greater or less than within surveyed areas. See Figure 1 for Region delineation and codes.

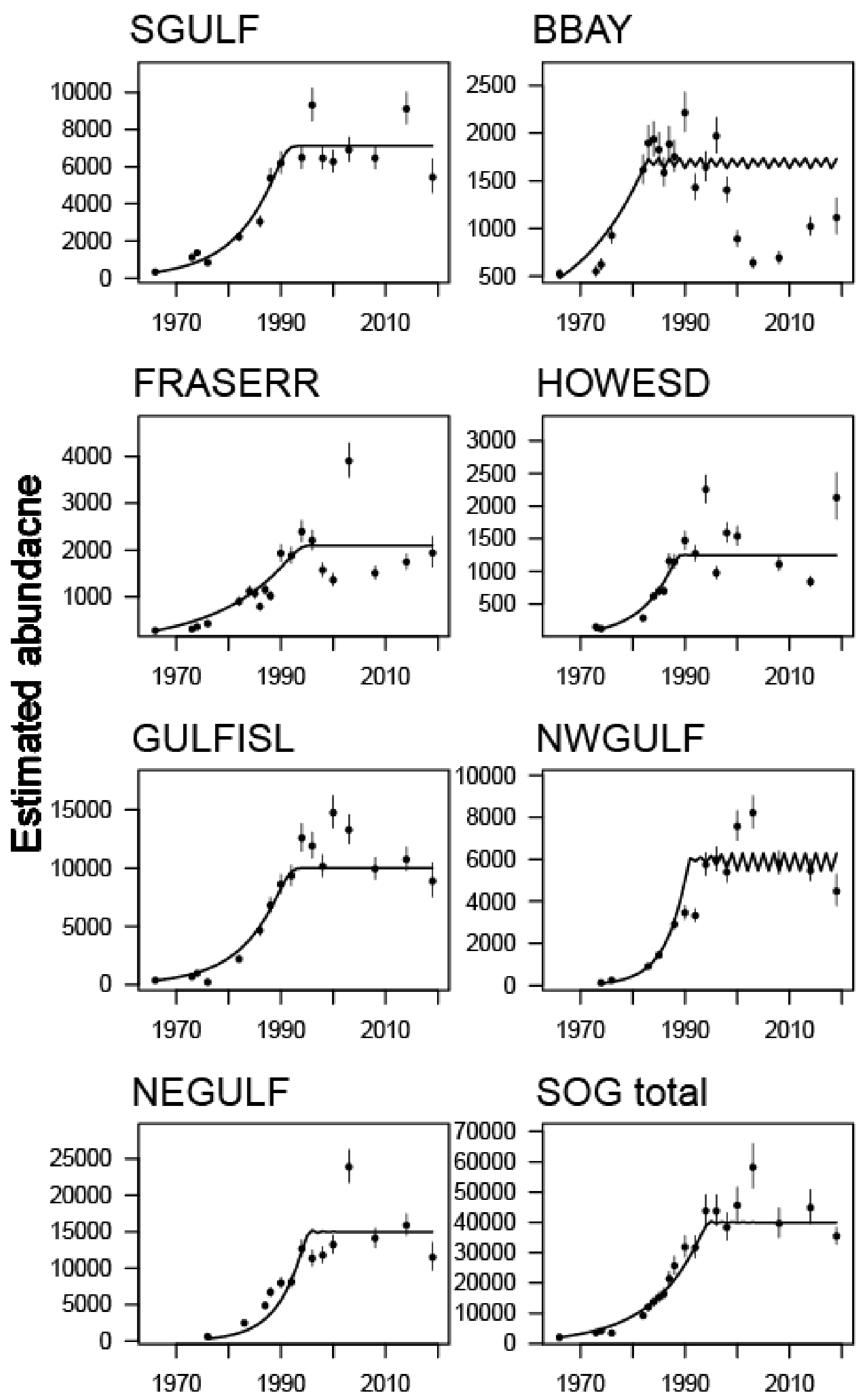

Figure 15. Abundance estimates (95% CI) of harbour seals for each subregion in the Strait of Georgia (SOG) and for the entire the SOG, with theta-logistic model fits. Abundance estimates were weighted by the proportion of shoreline covered. See Figure 1 for the delineation of the SOG and subregion codes. The plateau is typically interpreted as carrying capacity.

APPENDIX A. DFO AREA SHORELINE LENGTH REVIEW

Olesiuk (2010) adjusted seal counts for the un-surveyed shoreline by standardizing counts to the shoreline lengths in DFO Pacific fisheries management areas (PFMA). However, while this GIS layer delineates boundaries between DFO Pacific fisheries management areas, it is an incomplete representation of the coastline. Below we demonstrate step by step how this conclusion was reached and outline the assumed steps executed by Olesiuk (2010; PFO) and associated data files, contrasted with the steps undertaken in this document by the DFO Pinniped Research Program (PRP) to estimate shoreline length.

ISSUE

Observed mismatch between reported values in PFO 'digitizing' tab and current outputs

FILES IN USE

- 'Original' DFO pmfa shapefile (multi polygon layer defining the **waters** for each management area; no land even it appears there is a land layer)
- PRP created DFO pfma shapefile (multi polygon layer defining the **borders** of the management areas, no holes)
- CHS low water line (LWL; multiline layer defining the BC shoreline at low water)

BACKGROUND

PFO reported shoreline lengths of **DFO Management units** 1-29 in 2 columns (Olesiuk data file):

- 'Digitized' totaling ~ 38,000 km
- 'Km shoreline' where he took the proportion of the digitized length of each DFO area and standardized it to 27,200 km shoreline
- Why standardize? Why decide that the total shoreline of all DFO areas was actually 27,200 km and the digitized lengths needed to be corrected?
- Uncertain which files PFO had but the 'original' dfo pfma file has been available at PBS for > a decade

PRP calculated shoreline lengths of **PRP subregions**:

- Used CHS low water line file
- Clipped low water lines by PRP subregion boundaries
- Summed lengths of line segments for each subregion and then totaled for region

PMFA areas #3,5,6,7,8,9, & 10 share boundaries with PRP subregions in the NMC (PMFA-03, -05, -06) and the CMC (PFMA-07, -08, -09, -10)

- Allowed for almost direct comparison between PFO calculations and current PFO analysis
- Found good agreement between most PFO 'Digitized' lengths and PRP shoreline lengths for comparable areas (Table 1). However. This was **not** the case for PFO estimated 'KM Shoreline' (Table 1)
- Differences were in most cases minor and could easily be attributed to differences in input files, rounding by functions in various software employed, or both

- PMFA-09 was not a good match (1,200 digitized km by PFO but only 954 km by PRP)

Table A1. Estimated Olesiuk (PFO) and current study (PRP) shoreline lengths (km) for subset of DFO management areas. 'Km shoreline' is the proportion of the digitized length of each DFO area standardized to 27,200 km shoreline.

DFO Area	PFO estimations		PRP estimations	
Area number	Digitized Length (km)	km Shoreline	Subregion	Length km
3	1,666	1,175	PFMA_03	1,561
5	2,651	1,869	PFMA_05	2,679
6	4,023	2,836	PFMA_06	4,216
7	3,951	2,785	PFMA_07	3,974
8	1,695	1,195	PMFA_08	1,633
9	1,203	848	PFMA_09	954
10	611	431	PFMA_10	616

FOLLOW-UP

Similarities prompted PRP to compute shoreline lengths for all DFO management areas (Table 2).

- Used CHS low water line file
- Clipped low water lines by PRP created DFO pmfa boundary shapefile
- Again, generally good agreement between PFO 'Digitized' and PRP Lengths. Differences can mostly be assumed to be due to different inputs between PFO and PRP

Table A2. Estimated Olesiuk (PFO) and current study (PRP) shoreline lengths (km) for DFO management areas.

DFO Area	PFO	PRP
Area number	Digitized km	PMFA Length km
1	869	961
2	3626	3615
3	1666	1569

DFO Area	PFO	PRP
Area number	Digitized km	PMFA Length km
4	1759	1764
5	2651	2678
6	4023	4103
7	3951	3975
8	1695	1631
9	1203	955
10	611	616
11	1076	927
12	3592	3424
13	1674	1581
14	343	334
15	825	841
16	1031	995
17	583	574
18	479	485
19	405	420
20	325	261
21	51	51
22	71	12
23	1014	966
24	1235	1016
25	1107	1023

DFO Area	PFO	PRP
Area number	Digitized km	PMFA Length km
26	743	1032
27	846	860
28	567	612
29	559	646
Total	38,579	37,926

FURTHER, SO WHERE DID THE 27,200 KM STANDARDIZATION ORIGINATE? ARE WE MISSING AN IMPORTANT STEP?

In attempt to work out origins of PFO conversion to 'Km Shoreline', PRP clipped the low water polyline with the 'original' DFO PFMA polygon file that PFO likely had access to (Figure 1).

- File may seem suitable as an 'overlay' with which to clip areas but is somewhat deceiving
- File contains 'holes' in geometry where land would be and defines water only (Figure 2 and 3)
- Therefore, there would have to be 100% agreement between the water edges and the shoreline file in use in order for a geometric operation to obtain a valid shore length using this file

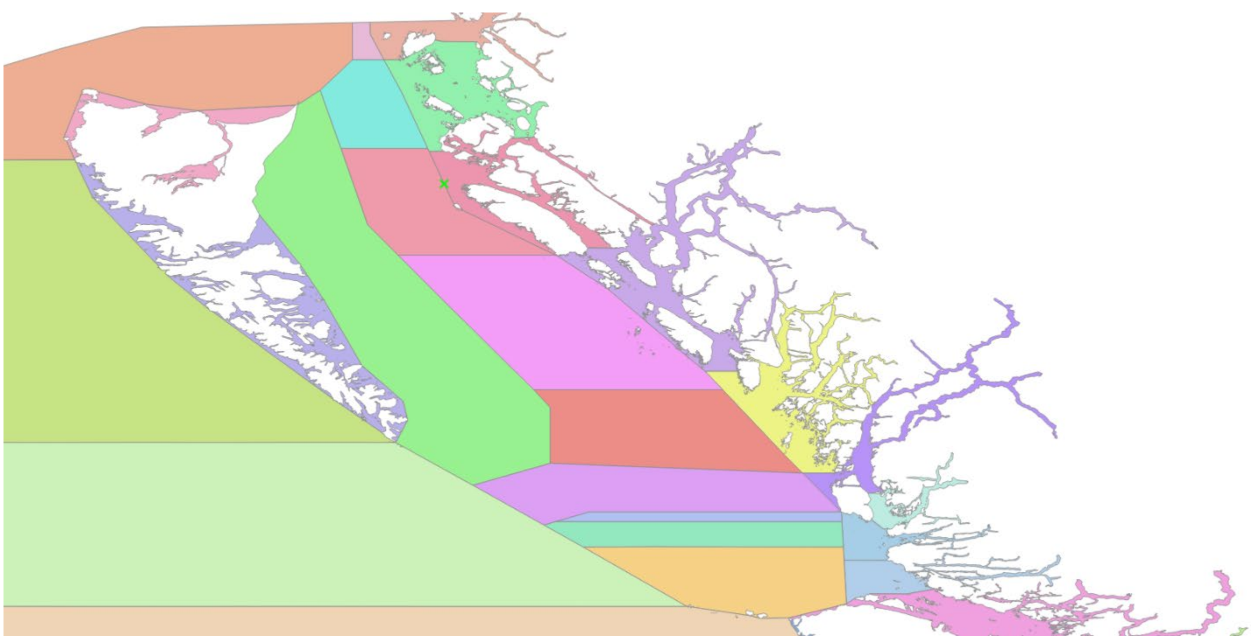

Figure A1. 'Original' DFO management area polygon file. Coloured shapes are different management areas. Note: white space is the absence of data and not shoreline.

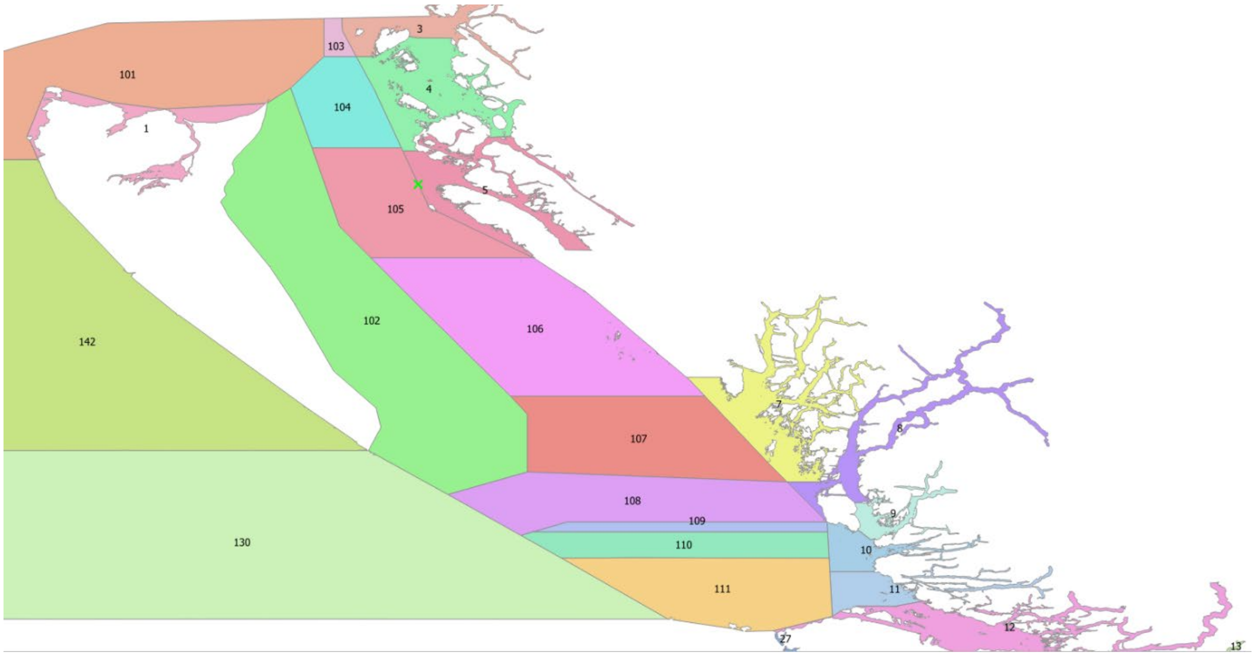

Figure A2. 'Original' DFO management area polygon file with Area 02 and Area 06 disabled to demonstrate that the polygons define water edges and not shore.

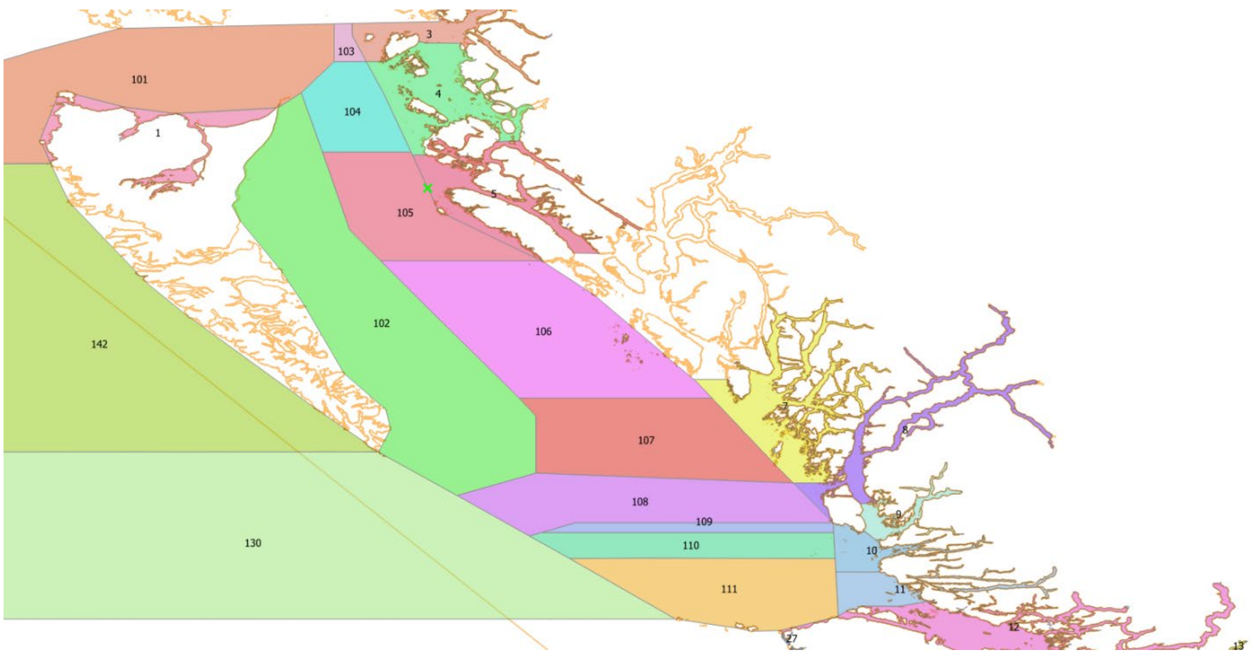

Figure A3. 'Original' DFO management area polygon file with Area 02 and Area 06 disabled and with the CHS low water line added to introduce a shoreline which can agree with the polygon edges.

- Subsequently, calculating lengths from the clip are similar to PFO 'Km Shoreline' (Table 3)
- Specific areas do vary but the overall tally is more similar to each other, summing to approximately 26-27,000 km, than to the ~38,000 km

Table A3. Estimated Olesiuk (PFO) and current study (PRP) shoreline lengths (km) for DFO management areas clipped to the low water polyline with the 'original' DFO PFMA polygon file.

DFO Area	**PFO**	**PRP**
Area number	Km Shoreline	PMFA Length (km)
1	613	607
2	2,557	2,302
3	1,175	846
4	1,240	1,484
5	1,869	2,205
6	2,836	2,657
7	2,785	2,746
8	1,195	930
9	848	526
10	431	408
11	759	431
12	2,533	2,092
13	1,180	937
14	242	316
15	581	555
16	727	548
17	411	519
18	337	381
19	285	361
20	229	214

DFO Area	PFO	PRP
Area number	Km Shoreline	PMFA Length (km)
21	36	45
22	50	10
23	715	727
24	871	757
25	780	717
26	524	756
27	596	647
28	399	437
29	394	523
Total	27,200	25,685

SO WHY DO YOU WE THINK WE'RE RIGHT? WHY IS ~38K OF COASTLINE MORE VALID THAN ~27K AS DETERMINED BY THE DFO AREA POLYGON?

- The 'overlay' DFO Area polygon file that PRP uses does not have any missing geometry where land and water join; rather it's a contiguous multipolygon file that matches the boundaries of the original (Figure 4)
- It does **not** rely on edge matching with a shoreline file to intersect it

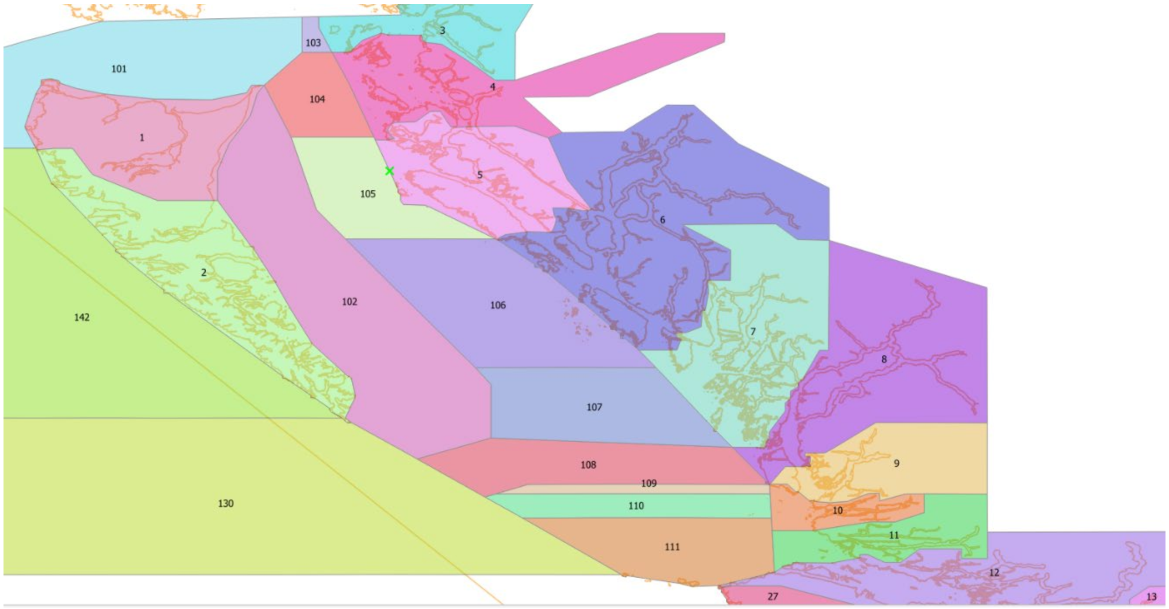

Figure A4. Contiguous PRP DFO management area polygon file with identical boundaries to original. CHS low water line visible beneath as an example of a feature that could be clipped by the overlaid DFO areas.

At coarse scales, there does not appear to be any issue with using the original DFO Area file (Figure 5 and 6).

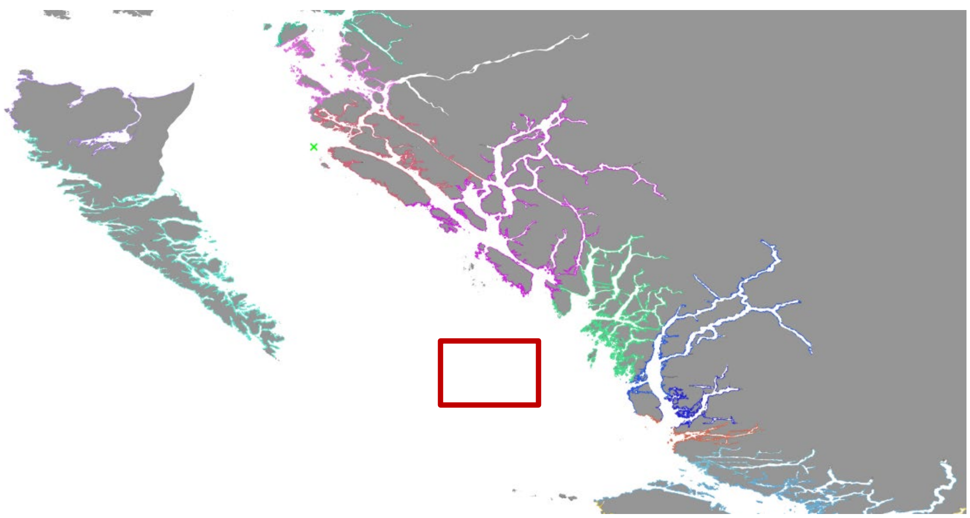

Figure A5. Shoreline clipped by original DFO management area polygon. Colours indicate shoreline assigned to different areas. The red box identifies Goose Is and Calvert Is and used in proceeding figures.

At finer scales, issues become more apparent as the water boundaries in the polygon did not fully align or overlap with the shoreline input file (Figure 7 and 8).

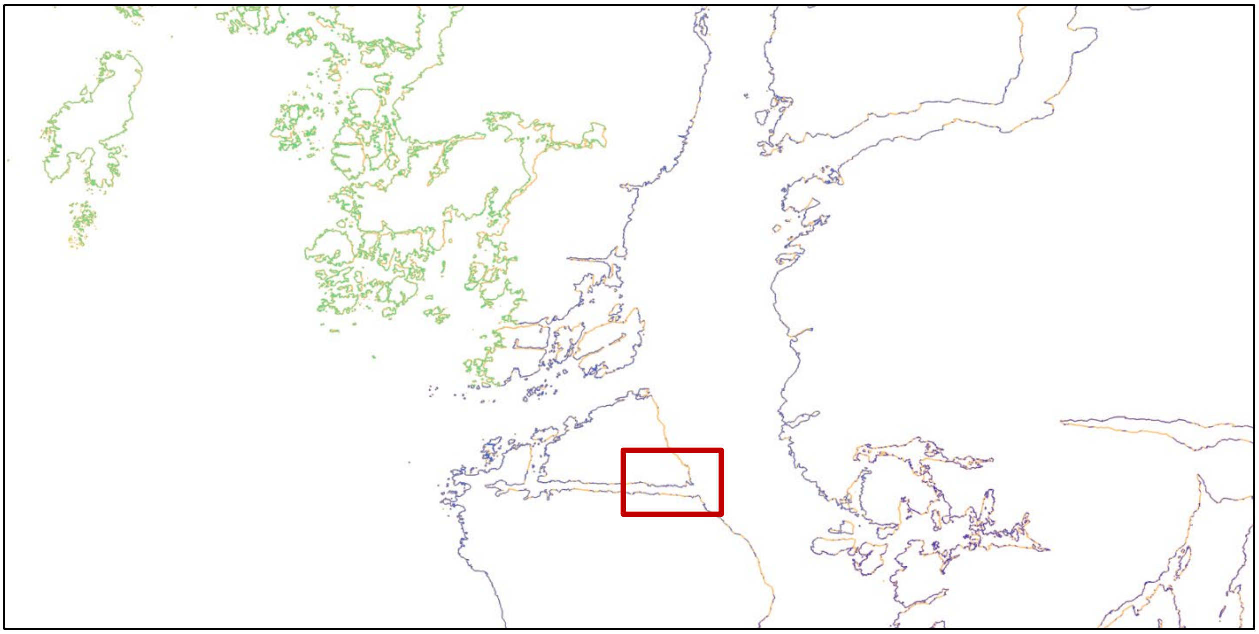

Figure A6. Shoreline clipped by original DFO management area polygon. Green colour indicates shoreline assigned to PFMA- 08 and Navy indicates PFMA-09. Orange is the low water line that was not clipped. The red box highlights the north end of Calvert Is and used in the proceeding figure.

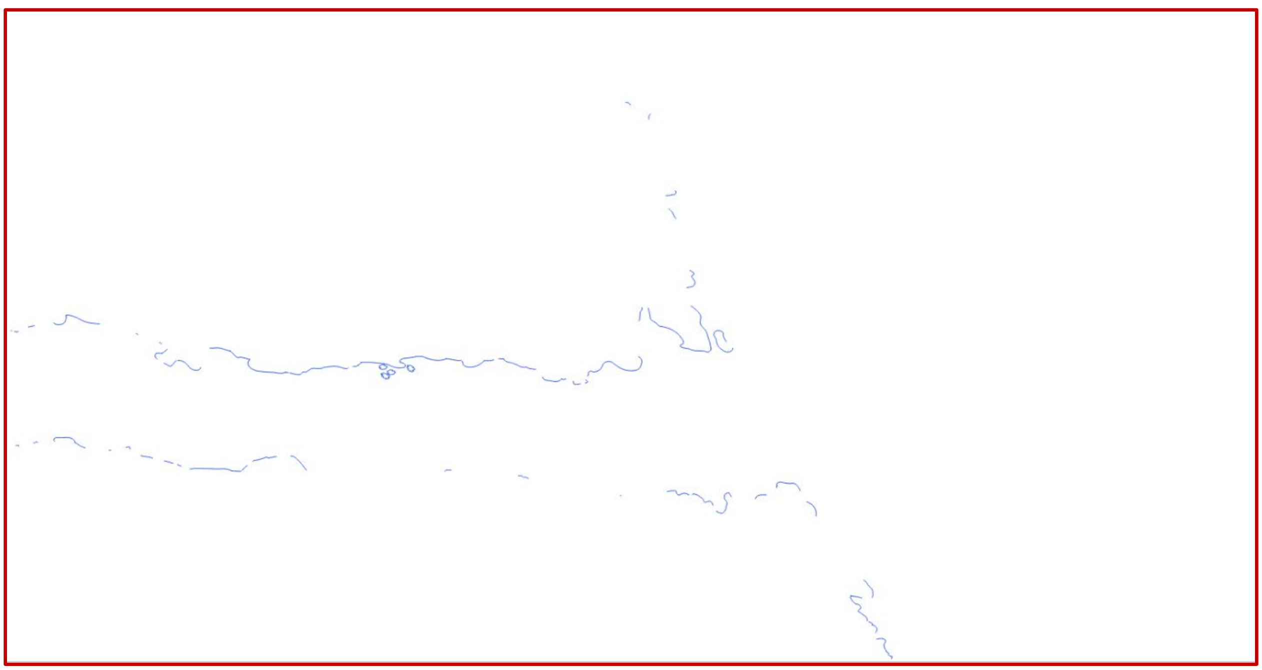

Figure A7. Shoreline from the northern end of Calvert I. clipped by original DFO management area polygon. Navy segments indicate lines assigned to PFMA-09. Note the numerous gaps in the line that were not assigned to PMFA-09 as the water boundaries in the polygon did not fully align or overlap with the shoreline input file.

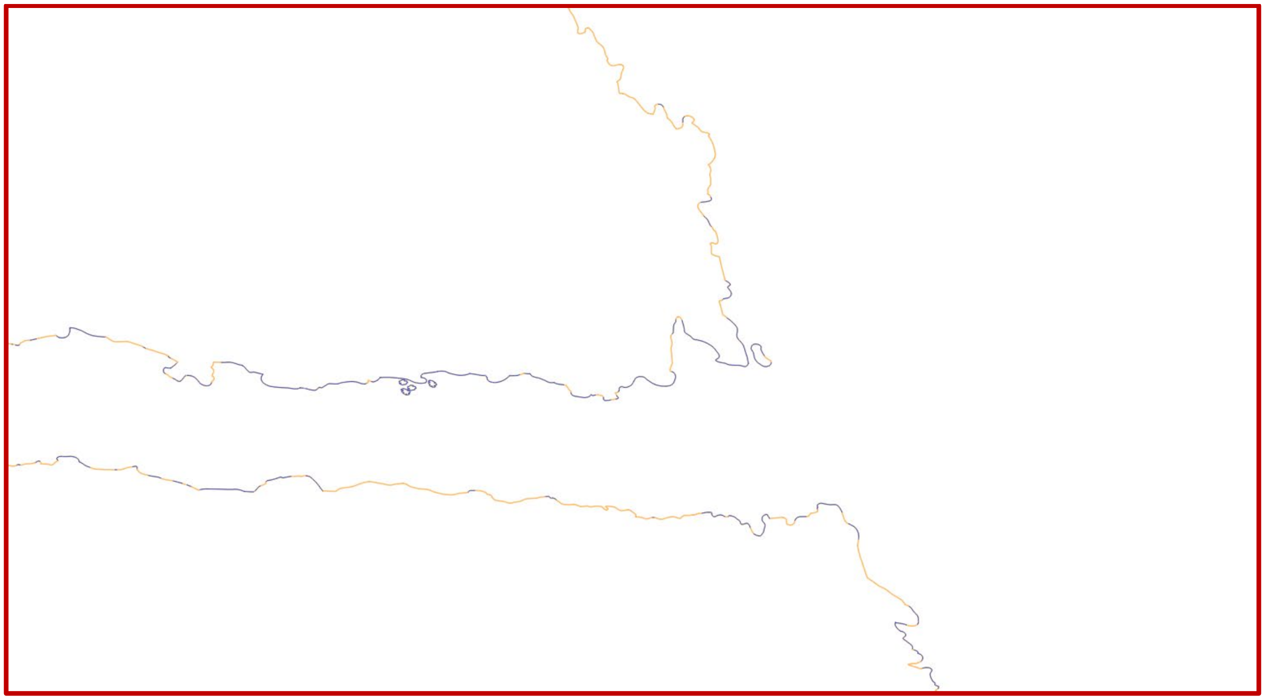

Figure A8. Shoreline clipped by original DFO management area polygon. Navy segments indicates shore is assigned to PFMA-09 and orange line is the CHS low water line added back-in to highlight shoreline that was not intersected and therefore not assigned to a management area.

Given the similar lengths in km generated from using the DFO Area polygon file in the past and during the current exercise, it's plausible PFO clipped a coastline file (similar to the low water line file employed here), with the original DFO PMFA file as a check against his 'Digitized' shoreline.

We surmise that he chose to scale or standardize his digitized lengths to the total shoreline length obtained from the original DFO file (27,200 km in his case) when he didn't find agreement with the earlier work.

We argue that shoreline estimates obtained using the DFO Management area polygons via an intersection or a 'clip' operation is not valid since substantial numbers of line segments go unassigned and therefore go unaccounted when shorelines lengths are calculated for DFO fisheries management areas.

We recommend using the Digitized line lengths for past work or substituting the PRP calculations obtained using a contiguous overlay and the CHS low water multiline spatial files as inputs.

REFERENCES CITED

Olesiuk, P.F. 2010. An assessment of population trends and abundance of harbour seals (*Phoca vitulina*) in British Columbia. DFO. Can. Sci. Advis. Secr. Res. Doc. 2009/105. vi + 157 p.

APPENDIX B. A BETA-BINOMIAL FRAMEWORK FOR COUNTS OF HARBOUR SEALS AT HAUL-OUT SITES

THEORETICAL FRAMEWORK

No correlation among individuals

If we assume that an individual seal spends an average proportion P of its time hauled-out, the probability that a single individual is hauled-out at any given time (e.g., available to be counted during a survey) is the result of a Bernoulli draw with probability P:

$$X \sim Bernouilli(P)$$

with X taking the value 1 with probability $\Pr(X = 1) = P$ and the value 0 with $\Pr(X = 0) = 1 - P$.

If this proportion P is the same for all N individuals at a given site, and if all individuals are independent of one another, then the number of individuals n available to be counted at any given time follows a binomial distribution:

$$n \sim Bin(N, P)$$

which has mean NP and variance NP(1-P).

Therefore, the variability in the number of hauled-out seals is expected to decrease quickly with increasing abundance (Figure B1). This low variability cannot account for what is observed in repeated counts of harbour seal haul-out sites and can result in unrealistic estimates of uncertainty around abundance estimates.

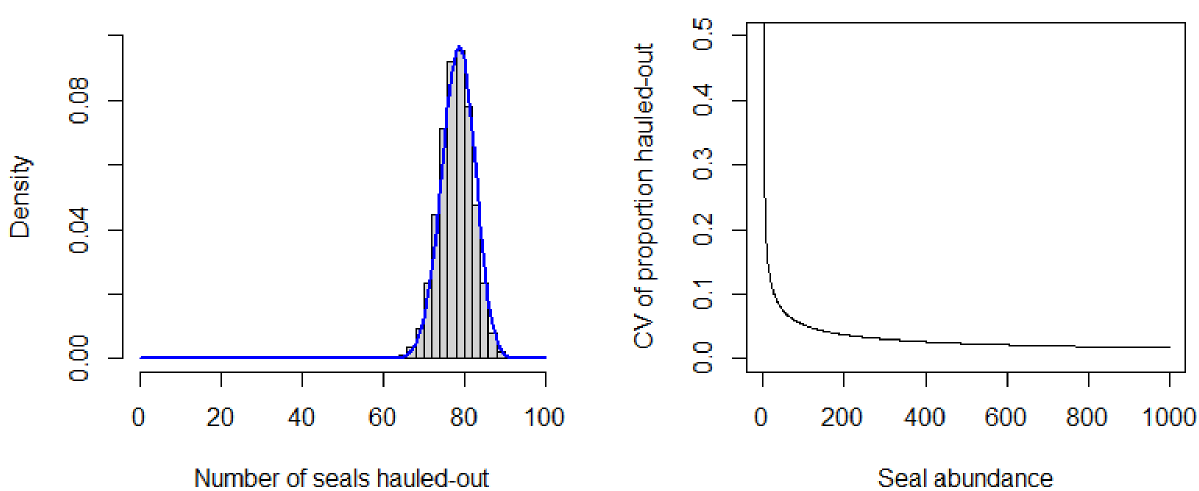

Figure B1. Left: Number of seals hauled-out for a site with abundance N=100 and P=0.785 under a binomial assumption. Histogram shows empirical distribution of 100,000 random draws from a binomial distribution. Blue line: probability density function of n ~ Bin(N,P). Right: Coefficient of variation of the proportion hauled-out as a function of seal abundance.

With correlated behaviour

Observations suggest that counts of hauled-out seals are more variable than expected under a binomial framework. This additional variance (i.e., overdispersion) can arise from heterogeneity in P among individuals in the population (e.g., sex, age classes, physiological state) or from correlation in haul-out behaviour among individuals. If seals are not independent from one another in their hauling behaviour, either because of social factors or because different individuals seek out the same environmental conditions (e.g., Lydersen et al. 2008), this results in higher variance than expected from a regular binomial distribution.

Under this assumption, the underlying Bernouilli trials are considered correlated and their sum (the number of seals hauled-out at any given time) can be modelled with a beta-binomial distribution (Skellam 1948):

$$n \sim BetaBin(N, P, \rho)$$

with ρ being the correlation factor among seal individuals.

This beta-binomial distribution has been shown to better fit the variability in observations of other correlated marine mammals such as walrus haul-out sites (Doniol-Valcroze et al. 2016) and manatee surveys (Martin et al.2011). In this framework, the variance of n is multiplied by an overdispersion factor $\sigma^2 = 1 + (N-1)\rho$. Therefore, the variability in numbers hauled-out is larger and the CV remains high even as abundance increases (converging to a value that only depends on P and ρ).

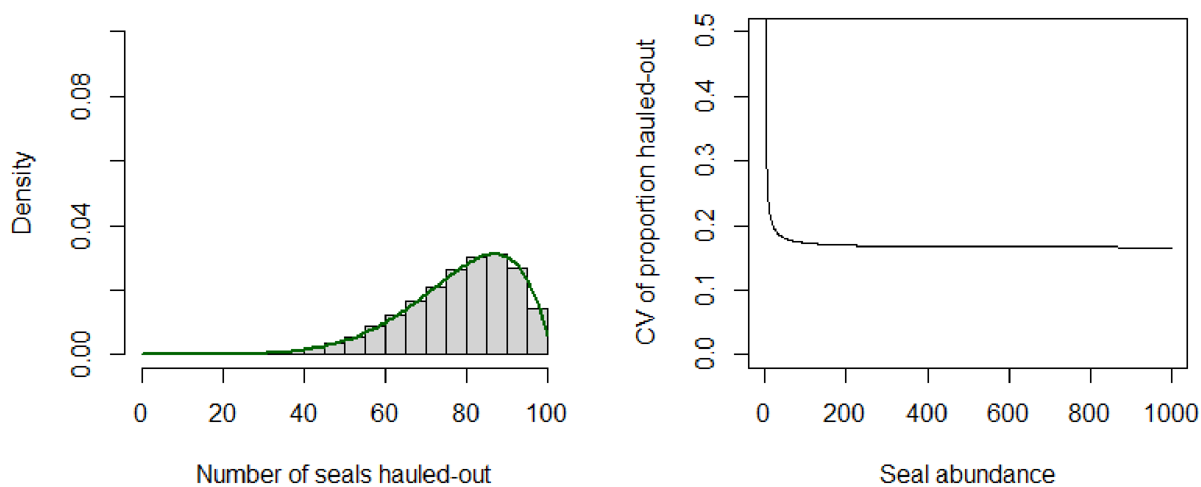

Figure B2. Left: Number of seals hauled-out for a site with abundance N=100, P=0.785 and ρ=0.10 under a correlated framework. Histogram shows empirical distribution of 100,000 random draws from a beta-binomial distribution. Blue line: probability density function of n ~ BetaBin(N,P, ρ). Right: Coefficient of variation of the proportion hauled-out as a function of seal abundance.

ABUNDANCE AND VARIANCE ESTIMATORS

No uncertainty around *P*

We assume that K repeated counts C1, C2, …, CK are made at a haul-out site with a true (unknown) abundance of N seals. At any given time, the number of seals hauled-out, n, follows a beta-binomial distribution with mean proportion P and correlation factor ρ. We also assume that these counts are made without error (i.e., $C_k = n_k$) and that P is known without error.

If $P < 1$, the mean of multiple survey counts $C_{mean} = \frac{1}{k}\sum_{\mathrm{k=1}}^{K} C_k$ will always be lower than N. However, if it is then corrected by a reliable estimate of P, it has been shown to be an unbiased estimate of N (Doniol-Valcroze et al. 2016). Thus, the estimator is:

$$\widehat{N} = \frac{C_{mean}}{P}$$

(eq.1)

and its variance is estimated as:

$$var(\widehat{N}) = \widehat{N}\,\frac{1-P}{k\,P} \times \sigma^2$$

(eq.2)

with:

$$\sigma^2 = 1 + (\widehat{N} - 1)\rho$$

(eq.3)

Uncertainty around the value of *P*

If there is uncertainty around the value of the haul-out proportion, P becomes a random variable with a normally-distributed error (SDP) around its point estimate $\hat{P}$. We assume that $\hat{P}$ and $var(\hat{P}) = SD_P{}^2$ are estimated from an independent study (e.g., telemetry).

With uncertainty around P, the estimator of abundance becomes $\widehat{N} = C_{mean}/\hat{P}$, which does not change its point estimate. To calculate its variance, we follow Thompson and Seber (1994) and use the delta-method approximation. The full variance becomes:

$$var(\widehat{N}) \cong \widehat{N}\,\frac{1-\hat{P}}{k\,\hat{P}} \times \sigma^2 + \frac{\widehat{N}^2}{\hat{P}^2}\,var(\hat{P})$$

(eq.4)

ESTIMATING CORRELATION COEFFICIENTS FROM TELEMETRY DATA

Estimating *ρ* from a sample of seals

When enough repeated counts are available from multiple sites, N-mixture models (Royle 2004) can be used to estimate P and ρ at the same time as N directly from the count data (e.g., Martin et al. 2011). However, due to costs and logistical challenges, pinniped aerial surveys are rarely repeated often enough to make this kind of inference possible. For walrus in the Canadian Arctic, Doniol-Valcroze et al. (2016) used telemetry data to estimate P as a first step, then used a series of repeated counts from an earlier time series at a single site to estimate ρ. These two parameters were then applied to the count data to estimate N.

Other options exist. When telemetry information is available from multiple individuals at the same time (i.e., concurrent deployments), correlation in their behaviour can be used to estimate ρ from the overdispersion measured in the data.

As an example, we simulate a group of N=30 seals that are tracked concurrently over 5,000 events using the R function rbetabinom, with P=0.785 and ρ=0.10 (Figure B3). Pretending we do not know any parameter other than P, we estimate abundance as $\widehat{N} = \frac{C_{mean}}{P} = \frac{23.671}{0.785} = 30.16$. We then measure the empirical variance of the observed counts var(C) = 19.07 and compare it with the expected variance for a binomial distribution, which is $NP(1-P)$ = 5.06. The ratio between the two gives us the overdispersion multiplier $\sigma^2 = 3.76$. In other words, the variance of these data is almost four times higher than expected under a binomial assumption.

From $\sigma^2 = 1 + (\widehat{N} - 1)\rho$, we calculate $\rho = (\sigma^2 - 1)/(\widehat{N} - 1)$ = 0.0954, showing that we have been able to estimate the value of ρ used to simulate the data in the first place (0.10).

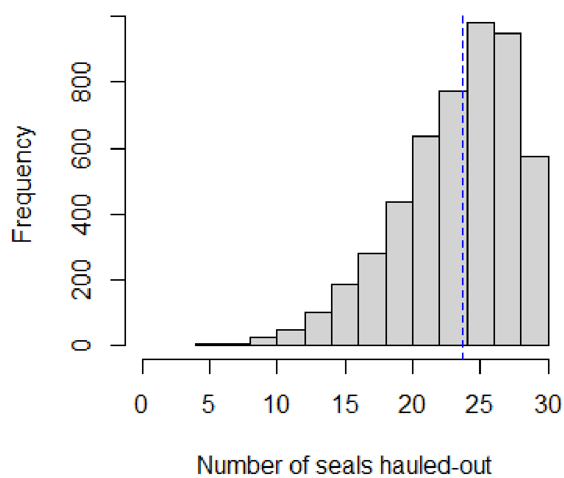

Figure B3. Empirical distribution of 5,000 draws from a beta—binomial distribution with N=30, P=0.785 and ρ=0.10, simulating the haul-out behaviour of 30 tagged seals. Dotted line is the mean of the observed counts.

Estimating ρ from multiple telemetry samples

Telemetry data seldom resemble the simulation presented above. The logistics of tagging free-ranging animals and the varying duration of each tag means that in most cases, the number of seals tagged concurrently will vary over time. Some tagging events will even take place in different years. To apply the method described above, one could limit the dataset to one particular period when the number of concurrently tracked individuals is constant. Such truncation, however, means a lot of data will be wasted even though they contain useful information. If one assumes that average seal haul-out behaviour does not change over time, even data from different years can be combined (i.e., a group of 5 tagged seals in one year is expected to behave the same as another of group of 5 years in a different year).

Since the above formula for ρ is a function of the sample size N, it cannot be applied directly to a dataset where N varies over time. Instead, we suggest assuming that the observed dataset is a mixture of m beta-binomial distributions (one for each sample size):

$$n_i \sim BetaBin(N_i, P, \rho)$$

with i = 1, 2, …, m. Each distribution has a different sample size $N_1, N_2, \dots, N_m$ but we assume that they have the same P and ρ parameter values.

The mixture distribution needs to be weighted for the relative contribution of each beta-binomial. For each sample size N_i, we sum the number of events zi or time-units that groups of seals of that size were tracked (combining all groups of seals of the same size across different time periods. We define the weights $w_i = z_i / \sum_1^m z_i$.

Assuming equality of means, the variance of the mixture distribution is simply the weighted sum of the variances of each beta-binomial. Expressed as the proportion of seals (rather than the raw counts), the variance for each distribution is:

$$var(P)_i = \frac{P(1-P)}{N_i}\ (1 - (N_i - 1)\rho)$$

(eq.5)

The variance for the weighted mixture distribution is:

$$var(P)_{mix} = \sum_1^m w_i \frac{P(1-P)}{N_i}\ (1 - (N_i - 1)\rho)$$

(eq.6)

Solving for ρ:

$$\rho = \frac{\frac{var(P)_{mix}}{P(1-P)} - \sum_1^m \frac{w_i}{N_i}}{\sum_1^m \frac{w_i (N_i - 1)}{N_i}}$$

(eq.7)

In other words, the correlation factor ρ can be estimated from the total variance in the observed dataset (that combines all the tagged individuals) if the number of events are known for each sample size. Simulations similar to those of 3.1 were performed to ensure that this estimator could find the true value of ρ for a large range of parameters.

A NOTE ON STRATIFICATION AND SUMMING MULTIPLE SITES

In a binomial framework, summing the counts made at J multiple sites is straightforward: the sum of J binomial distributions $n_1, n_2, \dots, n_J$ with sizes $N_1, N_2, \dots, N_J$ is itself a binomial distribution:

$$\sum_{j=1}^J n_j \sim Bin\left(\sum_{j=1}^J N_j, P\right)$$

with variance:

$$var\left(\sum_{j=1}^J n_j\right) = \left(\sum_{j=1}^J N_j\right) P(1-P) = \sum_{j=1}^J N_j P(1-P) = \sum_{j=1}^J var(n_j)$$

In other words, the variance of the sum of sites is the same as the sum of the variances of the sites. This means that under a binomial assumption, there is no difference between calculating the variance and CV by adding sites (i.e., as if each site was a different stratum) or by calculating the variance of the total abundance using the binomial formula.

In a beta-binomial framework, however, this equality does not hold. The reason for this can be explained intuitively using an extreme example: let us assume that seals haul-out 50% of the time and are correlated at $\rho = 0.95$, so that most of the time they are either all hauled-out or not at all. If one surveyed a site with 100 seals multiple times, the distribution of counts would made mostly of zero counts or counts of the full 100 seals; the mean would be around 50 but the variance would be extremely large (Figure B4a).

If instead the site was surveyed as 50 smaller sites with a true abundance of 2 seals on each, then for each small site, the counts would be mostly 0 or 2 (Figure B4b). If these 50 small sites were considered independent of one another and summed, the sum would be made of a mix of 0's and 2's (it would be extremely unlikely to have 50 0's or 50 2's) and the total count would have the same mean of 50 but a narrow distribution around the mean (Figure B4c).

In practical terms, this means that the scale at which the correlation among individuals is assumed must be considered carefully. Excessive stratification (i.e., assuming sites are independent if they are not) leads to low variance and is less conservative. Conversely, assuming that seals are correlated across the whole coast likely inflates the variance unrealistically.

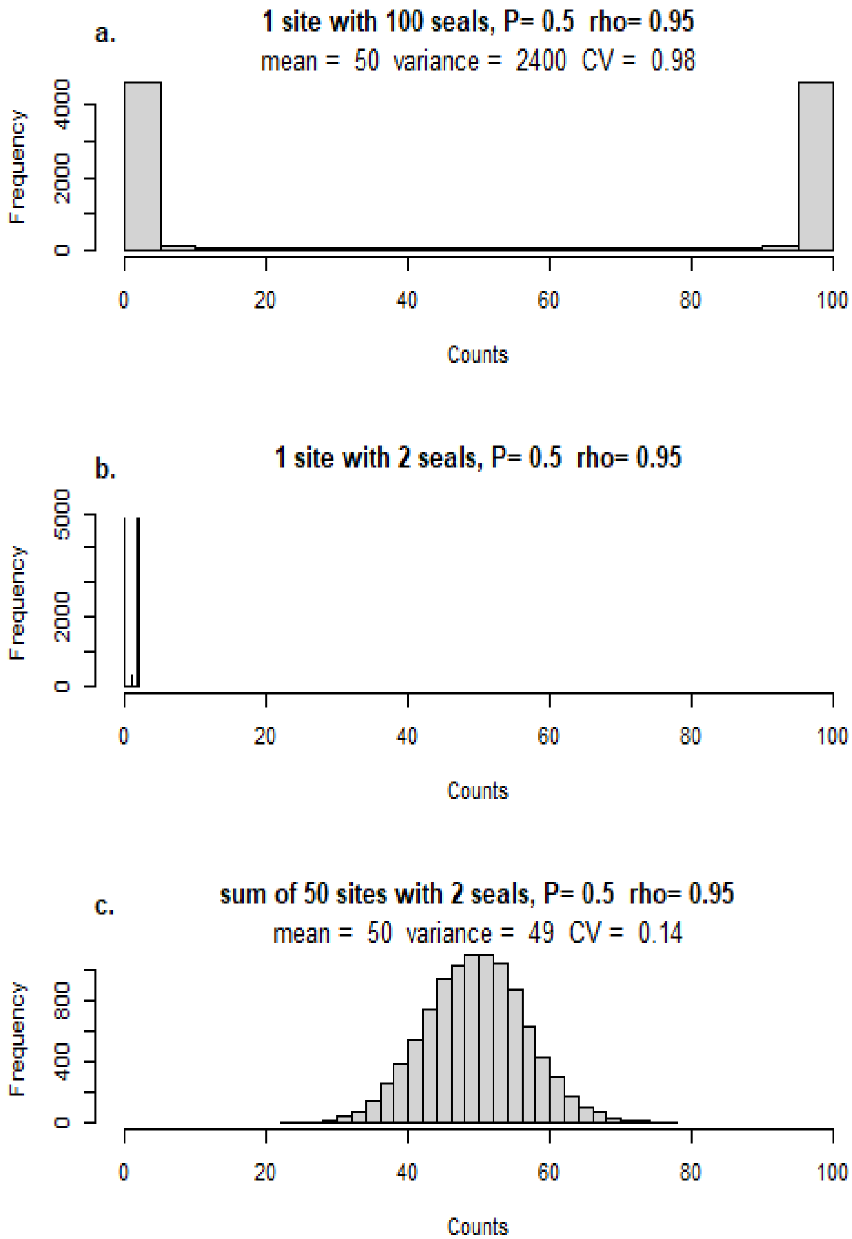

Figure B4. Frequency of 10,000 draws from a beta—binomial distribution showing the expected counts when surveying (a) a site with 100 seals, (b) a site with 2 seals, and (c) the sum of 50 sites with 2 seals.

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