Distribution and abundance of marine mammals in the coastal waters of British Columbia, Canada

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ABSTRACT

Information on animal distribution and abundance is integral to wildlife conservation and management. However abundance estimates have not been available for most cetaceans inhabiting the coastal waters of Canada's Pacific coast, including those species that were heavily depleted by commercial whaling. We conducted systematic sightings surveys in the Inside Passage waters between the British Columbia (BC)-Washington and the BC-Alaska borders, covering 4,400km (2,400nm) of trackline in the summers of 2004 and 2005. Province-wide abundance estimates (with 95% confidence intervals) assuming certain trackline detection for seven cetacean species are as follows: harbour porpoise, 9,120 (4,210-19,760); Dall's porpoise, 4,910 (2,700-8,940); Pacific white-sided dolphin, 25,900 (12,900-52,100); humpback whale, 1,310 (755-2,280); fin whale, 496 (201-1,220); minke whale, 388 (222-680); and 'northern resident' killer whale, 161 (45-574). We also calculated density of harbour seals, and estimate that total abundance of harbour seals in the study area was 19,400 (14,900-25,200). To the best of our knowledge, these are the first abundance estimates for this region for each of these cetacean species except killer whales. Small sample size makes our killer whale estimate tenuous, but one worth noting, because it is close to the known number of northern resident killer whales (2004 census was 219 animals, Pacific Biological Station, DFO, Canada). Our minke whale abundance estimate is similarly tentative, however our results do indicate that minke whales were rare in this region. Minke whales were recently deemed to be *Not at Risk* in Canada's Pacific region by the Committee on the Status of Endangered Wildlife in Canada, but our abundance estimates suggest that minke whales were just as rare as the *Threatened* fin, humpback and killer whales. Our results provide a province-wide estimate of the proportion (roughly two-thirds) of harbour seals that were in the water on average at any point during our survey, while one-third were hauled out. While the majority of harbour seals were found as expected in the southern straits and in the mainland inlets, a substantial number of animals were on the north coast and in the Queen Charlotte Basin as well. These data also provide the first systematic snapshot of summertime distribution and abundance of marine mammals in the Queen Charlotte Basin, where offshore oil and gas development is being proposed. These data should provide a basis for risk assessment of seismic surveys on acoustically sensitive mammals. Similarly, our abundance estimates should be used to assess sustainable limits to incidental by-catch in commercial fisheries.

KEYWORDS – SURVEY, NORTH PACIFIC, SYSTEMATIC, LINE TRANSECT, MULTI-SPECIES, ABUNDANCE, DISTRIBUTION

INTRODUCTION

Marine mammals in coastal British Columbia

More than 20 marine mammal species can be found in the coastal waters of British Columbia (BC), Canada. These species vary widely in their fidelity to inshore Canadian waters, history of exploitation, conservation status, and the extent to which they have been studied. Killer whales (*Orcinus orca*) in BC are among the best-studied cetacean populations anywhere (Bigg *et al*. 1990; Olesiuk *et al*. 1990; Ford *et al*. 1998, 2000). Baleen whales were the subject of extensive pelagic and coastal whaling in the northeast Pacific; the last coastal whaling stations in BC closed in 1967 (Gregr *et al*. 2000). Incidental by-catch in commercial gillnet fisheries does occur (Hall *et al*. 2002). Whalewatching, once seen as an alternative to whaling, is now considered a potential threat (Williams *et al*. 2002ab) to some cetacean populations via masking effects of boat noise, potential energetic cost of vessel avoidance tactics and disruption of feeding activity, and emission of outboard motor exhaust. All marine mammals in BC, with their remarkable acoustic sensitivity and high trophic position, are vulnerable to impacts of intense anthropogenic noise and toxicity of fat-soluble contaminants. In recent years, there has been considerable discussion about lifting the existing moratorium on offshore oil and gas exploration and extraction off the north and central coasts of BC, which has created a heightened sense of urgency to collect baseline data on marine mammal distribution and abundance.

The following is a summary of frequently seen cetacean and pinniped species in BC's inshore waters. With few exceptions, our knowledge of populations reflects our pattern of use of that species. Exploited populations (either for hunting, live capture, culling, or non-consumptive uses such as whalewatching), receive much greater scientific attention than unexploited ones. The species' status in BC refers to that determined by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC, http://www.cosewic.gc.ca). COSEWIC uses a variety of information sources to assess extinction risk for species, and to report its assessment to the Canadian government and the public. At that point, species that have been designated by COSEWIC may or may not qualify for legal protection and recovery efforts under Canada's Species at Risk Act (SARA).

Cetaceans

Harbour porpoise (*Phocoena phocoena*) are listed as a species of ***Special Concern*** in Canada's Pacific waters (COSEWIC 2003a). Anthropogenic activity, pollution and by-catch have been flagged as conservation threats (Baird 2003a). Rates of by-

catch were estimated by Hall *et al.* (2002), and studies on harbour porpoise habitat usage are being studied off southern Vancouver Island (Hall 2004). No province-wide estimates are currently available for abundance or trends in abundance.

Dall's porpoise (*Phocoenoides dalli*) are thought to be ***Not at Risk*** in BC (Jefferson 1990). They are widely distributed, and commonly seen in deep coastal waters, but no province-wide estimates are available for abundance or trends in abundance.

Pacific white-sided dolphin (*Lagenorhynchus obliquidens*) are thought to be ***Not at Risk*** in BC (Stacey and Baird 1991). This species appears to have returned to inshore waters in BC after a decades-long absence (Heise 1996, Morton 2000), and was called the most abundant cetacean in this region (Heise 1996). Interactions with fisheries are rare locally (Hall *et al*. 2002), and in other areas of the North Pacific (SAR 2000). No province-wide estimates are available for abundance or trends.

Minke whale (*Balaenoptera acutorostrata*) status is currently in review by COSEWIC. The species was never targeted by coastal or pelagic whalers in western Canadian waters. There is some evidence that individual minke whales may be resident to inshore coastal waters of Washington state (Dorsey *et al*. 1990), but the species is poorly studied in BC waters. No province-wide estimates are currently available for abundance or trends in abundance.

Humpback whales (*Megaptera novaeangliae*) are listed as ***Threatened*** in Canada's Pacific waters (COSEWIC 2003b). They were reduced to a fraction of pre-exploitation numbers by commercial whaling (Baird 2003b), but there is strong evidence to suggest that the North Pacific population is recovering (Calambokidis *et al*. 1997). Photo-identification on animals that use British Columbian waters is extensive, collaborative and ongoing (coordinated by Fisheries and Oceans Canada (DFO)).

Fin whales (*Balaenoptera physalus*) are listed as ***Threatened*** in Canada's Pacific region (COSEWIC 2005). The species was heavily exploited by commercial whaling, with evidence suggesting that the population was hunted to near commercial extinction by the 1960s (Gregr *et al*. 2000, Gregr and Trites 2001). Ship strikes and fishing gear entanglement are potential threats to fin whale recovery in BC. No province-wide estimates are currently available for abundance or trends in abundance.

Two fish-eating (*i.e.*, northern and southern 'resident') populations of killer whales (*Orcinus orca*) inhabit the coastal waters of BC, as do a mammal-hunting 'transient' population and a recently discovered and poorly studied 'offshore' population (Ford *et al*. 1998, 2000). Not only is abundance known for the fish-eating killer whales, but also it is known with an unusually high degree of confidence (Ford *et al*. 2000). Absolute abundance of mammal-hunting killer whales is more difficult to estimate than that of fish-eating killer whales, because the strong differences in social structure make it difficult to choose appropriate capture-recapture statistical models for transient killer whale photo-identification data. Northern resident and transient populations are considered ***Threatened*** in BC waters, while the southern resident population is listed as ***Endangered***. The offshore population is considered to be of ***Special Concern***. Conservation threats to the species in BC waters include: small population size due to a previous live-capture fishery for display (Bigg and Wolman 1975); anthropogenic noise; contaminants (Ross *et al*. 2000); and prey availability (Baird 2001a).

Other: The sei whale (*Balaenoptera borealis*), blue whale (*Balaenoptera musculus*) and North Pacific right whale (*Eubalaena glacialis*) are all listed as ***Endangered*** in BC, due primarily to historic overexploitation (Gregr *et al*. 2000) and, consequently, small current population size. The gray whale (*Eschrichtius robustus*) is listed as a species of ***Special Concern*** in BC (COSEWIC 2004). The deep-diving and relatively poorly studied beaked whales (Family Ziphiidae) are rarely reported in BC waters, although they may be more common than the scarce sightings would suggest (Willis and Baird 1998).

Pinnipeds

The harbour seal (*Phoca vitulina*) is considered to be ***Not at Risk*** in western Canada, due to large and increasing population size (Baird 2001b). Conservation concerns include prey availability and illegal and unreported shooting, or mortality incidental to fish farming operations (Baird 2001b). Their widespread distribution makes them less vulnerable to oil spills than those species that haul out in few locations. Their tolerance of urbanised habitat lends them susceptible to bioaccumulation of contaminants. DFO conducts regular counts of pinnipeds at haul-out sites and corrects for animals likely to be at sea, so trend data are available, particularly in southern BC, however density of harbour seals on the north and central BC coasts is less well studied, and at-sea distribution is poorly studied in BC waters generally (Olesiuk 1993).

Steller sea lions (*Eumetopias jubatus*) are listed as a species of ***Special Concern*** in western Canada (COSEWIC 2003c). While the species is locally abundant and growing, the breeding population in BC waters is composed of only three known breeding sites (COSEWIC 2003c), which makes them inherently vulnerable to catastrophic events such as oil spills. At-sea distribution of Steller sea lions is not well studied in BC waters.

Northern elephant seals (*Mirounga angustirostris*) are considered to be ***Not at Risk*** in BC waters. The population was hunted to near extinction in the 19th century, but the surviving population has increased exponentially since then. At-sea distribution of elephant seals is not well studied in BC waters.

Systematic surveys of Canada's Inside Passage
We conducted a combined visual and acoustic survey to achieve three primary goals with respect to marine mammals. The primary objective of the survey was to generate design-unbiased estimates of abundance and distribution of marine mammal species in BC coastal waters during the summer months. The second goal was to provide unbiased estimates of at-sea distribution. Here we report results from our systematic survey of marine mammals of the Inside Passage, which we completed in summers 2004 and 2005.

METHODS

Survey design
Our survey design is described in detail in a manuscript by Thomas *et al*. (this meeting). We used a stratified survey design, where the study area was divided into four strata, within which a sample of equal-spaced zig-zag or parallel transect lines was placed with a random start point to ensure equal coverage probability within strata.

1. Queen Charlotte Basin – this roughly convex region has been proposed for offshore oil exploration and extraction. For our purposes, it extends to a maximum of 82nm offshore in the west to an eastern boundary line drawn down the edges of the outer coastal islands, east of which we considered to be part of a mainland inlet stratum (see Stratum 4). The southern boundary was the narrow neck of Johnstone Strait, at the west end of West Cracroft Island.
2. Strait of Georgia and Juan de Fuca Strait – Canadian waters off southern Vancouver Island to the BC mainland shore.
3. Johnstone Strait and Discovery Pass – narrow passageway between northeastern Vancouver Island and mainland BC.
4. Mainland inlets – this collection of fjords, passages, straits and inlets was grouped using a GIS into 33 irregular-shaped bodies of water that could be surveyed in 1-3 days. From these 'primary sampling units' (PSUs), a sample of five was selected using a systematic random design, with probability of sampling proportional to area. Within each of these, a systematic parallel line design was used to generate transect lines. This provided a 10-day, cluster sample of the mainland inlets, which was designed to provide a reasonable starting point to represent the mainland inlet stratum.

Stratum 1 (Queen Charlotte Basin), was surveyed twice, in summers 2004 and 2005. The 2005 survey design for Stratum 1 was similar to 2004, but used a new, random starting point. The effort and coverage probability were similar in both years.

Field methods to measure animal density
Visual search effort
Data were collected aboard the sailing vessel, *Achiever* (a 21m steel-hulled sailboat) in 2004 and the *Gwaii Haanas* (a 20m aluminum power boat) in 2005. Data were collected from the highest accessible point (the *primary observer platform*) on the ships used in this study, such that eye height was approximately 5m. An aluminum platform was built for *Achiever* to increase the eye height of observers well above the ship's boom. The team consisted of six people. Three people served on the primary observer team, namely a port and starboard observer and a data recorder. In addition, one observer operated the computer while two team members were on rest periods.

The primary observer team searched ahead of the ship, that is, a sector from the trackline to 90˚ abeam the ship, while concentrating primarily on the trackline. Each observer searched a sector spanning from 30˚ on one side of the trackline to 90˚ on the other side. The data recorder recorded whenever a sighting was made, and assisted the observer with species identification or to group size estimation, when needed.

A GPS was connected to a computer running Logger software (www.ifaw.org). This collected positional information every 10s, which was used for calculating length of trackline covered, as well as ship's course and speed. The computer operator entered information on sighting conditions every 15 minutes, or as conditions changed. The computer operator also noted the position of each team member at the beginning of every hour. Observer rotation occurred every hour. The computer operator also recorded information on factors that could affect sighting conditions, including sea state, cloud cover and precipitation, and a subjective sightability code.

Sightings
Whenever a cetacean or school of cetaceans was spotted, it was assigned a sighting number and reported to the data recorder on the bridge via two-way radio. An angle board mounted on the deck railing was used to measure radial angle to the school, and a measurement was made of the range to the first sighting using 7X50 reticle binoculars or a graduated sighting gauge. If a visual estimate had to be made, then those radial distance estimates were corrected subsequently using observer-specific distance estimation experiments (described for similar experiments in Williams 2003). No attempt was made to assess whether range or angle *measurements* were biased; only visual *estimates* of range were corrected using these calibration experiments. The computer operator and data recorder noted ship location and the time of the sighting, and binoculars were used to confirm

species and school size. Additional information was recorded on cue type (*inter alia* body, blow, seabird activity), the animal's behaviour, and its heading relative to the ship.

Occasionally, a decision would be made to 'close' on a sighting. When the ship left the trackline, search effort was terminated. 'Closing' occurred usually when a large balaenopterid was not identified to species with certainty, but they also allowed collection of identification photographs of humpback and killer whales, and to obtain more accurate estimates of school size. After closing on a sighting, search effort was resumed once the ship reached cruising speed and rejoined the original trackline. We occasionally stayed 'on-effort' during transit legs to increase the number of sightings available for fitting the detection function. These sightings are subsequently referred to as transit-leg sightings and were not included in density estimates.

Data analysis

Standard analysis of line transect data (Buckland *et al.* 2001) can be split into three parts: (i) fitting a detection function to observed distances of sightings from the transect to estimate average probability of detection, p (Buckland 1985); (ii) using observed school sizes to estimate mean cluster size in the population, E(s); (iii) estimation of animal density, D, using the formula

$$\hat{D} = \frac{n\hat{E}(s)}{2wL\hat{p}} \qquad (1)$$

where n is the number of animals seen, w is the truncation distance and L is the total length of the transects. We deal with each of these parts below.

Data for each species were treated separately. Estimation of detection probability and mean school size was performed using the free software Distance 5.0 Beta 5 (Thomas *et al*. 2005). Distance was not used to estimate overall density or abundance due to the non-standard stratification used, so this was done using free statistical software R (R development core team 2005), version 2.2.0. Distance projects containing the data and analyses, and R code are available on request from the first author.

Estimation of detection probability

Let $g(x)$ be the probability of detecting a cetacean school at perpendicular distance x from the transect line. Schools greater than the truncation distance, w, from the line are excluded. When transect lines are located randomly within each survey stratum, the true distribution of animals is uniform with respect to the transect line. Therefore, the average probability of detection, p, is given by the average of g(x) between 0 and w, i.e.:

$$p = \int_0^w g(x)dx \Big/ w$$

Buckland *et al.* (2001) provide a framework for estimating $g(x)$ by fitting smooth, flexible functions to the observed perpendicular distances. We refer to the approach detailed in this book as 'conventional distance sampling' (CDS). We used Distance to fit the uniform, half-normal and hazard-rate key functions, together with polynomial or cosine series expansion terms as required. Model selection was guided by Akaike's Information Criterion (AIC) and by visual inspection of diagnostic plots, and selected models were judged using goodness-of-fit tests (Buckland et al. 2001, Section 2.5).

The reliability of the estimate of p is improved often by removing (truncating) the 5-10% most distant sightings before fitting the detection function, especially where no truncation was performed in the field (Buckland *et al.* 2001). Alternatively, a truncation distance, w, can be chosen so that $g(w) \approx 0.15$ (Buckland *et al.* 2001). We used Distance to assess the need for truncation, by fitting detection functions to data with 0, 5 and 10% of the most distant sightings removed and choosing the least amount of truncation necessary to achieve the same number of series expansion terms as were fit at 10% truncation (fitting fewer series terms will tend to give better precision) while keeping $g(w) > 0.10$. We then rounded the truncation distance to the nearest 100 meters. In general, the amount of truncation used had very little effect on the results.

Because our survey design involved stratification, we investigated the possibility of fitting a separate detection function in each stratum for species where sample size was sufficient (> 60 observations per stratum). The stratified detection functions were used if they had a lower AIC than the pooled function.

We also analyzed data collected for one species of pinniped, the harbour seal. Since we anticipated that there would be a significantly different detection function for seals seen on land compared with those in the water, we tried fitting detection functions stratified by land/water, and used these if they had a lower AIC than the pooled function.

Estimation of population mean school size

A common phenomenon in cetacean surveys is for the simple mean of the observed school sizes to be an over-estimate of the population mean, because large schools are more detectible at long ranges than small schools (Buckland *et al.* 2001, Section

3.5). We used the default method in Distance to obtain an unbiased estimate of population mean school size, as follows. A least squares regression was fit of the natural logarithm of school size (ln(s)) on the estimated probability of detection at the distance the school was seen. The predicted value of ln(s) at zero distance (when detection probability is 1) was then back-transformed (Buckland *et al.* 2001, formula 3.64) to provide the required estimate.

For harbour seals, we expected the cluster size to be different for observations in the water and seals that were hauled out, so we analysed the two groups separately.

Density, abundance and variance estimation

For each species, we estimated density, abundance and associated measures of uncertainty within each stratum, and then combined these estimates to produce results for the whole study area. For harbour seals, we also estimated these statistics separately within each stratum for seals on land versus those in the water. The methods used are an application of those given by Buckland *et al.* (2001, Section 3.6), but are slightly more complicated due to the stratification used in the survey design and the cluster sample in stratum 4. For comparability with Buckland et al. (2001), we give formulae for estimating density and note that abundance is simply density multiplied by area. Note also that the exact methods for obtaining the overall variance depend on the level at which the detection function and cluster sizes are estimated independently. In any event, the selected detection functions were and cluster sizes were fit to all data pooled, so we describe that scenario here. We deal with each stratum in turn, before considering the overall density estimate.

For stratum 1, let D_{1j} be the density of the species under consideration at the time of the survey in year j where j=1 (for 2004) or 2 (for 2005). Applying equation (1) and assuming that probability of detection (p) and population mean school size ($E(s)$) are estimated for all data pooled:

$$\hat{D}_{1j} = \frac{n_{1j}\hat{E}(s)}{2wL_{1j}\hat{p}} \quad (2)$$

where n_{1j} is the number of animals seen and L_{1j} is the total length of the transects in stratum 1 in year j. Let D_1 be the average density at the time of the survey, averaged over both years. This can be estimated by the mean of the year-specific estimates, weighted by survey effort to maximize precision:

$$\begin{aligned}\hat{D}_1 &= \frac{L_{11}}{L_1}\hat{D}_{11} + \frac{L_{12}}{L_1}\hat{D}_{12} \\ &= \frac{\hat{E}(s)}{2w\hat{p}}\left(\frac{L_{11}}{L_1}M_{11} + \frac{L_{12}}{L_1}M_{12}\right) \\ &= \frac{\hat{E}(s)M_1}{2w\hat{p}}\end{aligned}$$

where $L_1 = \sum_{j=1}^{2} L_{1j}$ and $M_{1j} = n_{1j}/L_{1j}$, the encounter rate (Note that these expressions can be simplified but we express them this way for generality). An estimate of the variance of $\hat{D}_1$ can be obtained using the delta method, and is

$$\hat{\text{var}}(\hat{D}_1) = \hat{D}_1^2\left(\frac{\hat{\text{var}}(\hat{E}(s))}{\hat{E}(s)^2} + \frac{\hat{\text{var}}(\hat{p})}{\hat{p}^2} + \frac{\hat{\text{var}}(M_1)}{M_1^2}\right) \quad (3)$$

where

$$\hat{\text{var}}(M_1) = \left(\frac{L_{11}}{L_1}\right)^2 \hat{\text{var}}(M_{11}) + \left(\frac{L_{12}}{L_1}\right)^2 \hat{\text{var}}(M_{12})$$

and $\hat{\text{var}}(M_{1j}) = \hat{\text{var}}(n_{1j}/L)$,

For strata 2 and 3, let D_i be the density in stratum i (i=2, 3), estimated by

$$\hat{D}_i = \frac{\hat{E}(s)M_i}{2w\hat{p}} \quad (4)$$

where $M_i = n_i/L_i$. The corresponding variance, $\hat{\text{var}}(\hat{D}_i)$, is estimated just as for equation (3), replacing the subscripts 1 with i.

For stratum 4, the formulae are very similar to stratum 1, except that we estimate overall density in the stratum as the simple mean of the estimates of density in each PSU, rather than the effort-weighted mean. Let D_{4j} be the density estimate in PSU j

in stratum 4, with j=1, … , 5 corresponding to the sampled PSUs 4, 10, 17, 21 and 29 respectively. D_{4j} can be estimated using an equation like (2), replacing the subscripts 1j with 4j. Average density for the stratum, D_4, can then be estimated using equation (4), where i=4 and $M_4 = 1/5\sum_{j=1}^{5} M_{4j}$ and $M_{4j} = n_{4j}/L_{4j}$. The corresponding variance, $\hat{\text{var}}(\hat{D}_4)$, can be estimated just as for equation (3), replacing the subscripts 1 with 4, and using $\hat{\text{var}}(M_4) = (1/5)^2 \sum_{j=1}^{5} (\hat{\text{var}}(M_{4j}))$.

Let D be the mean abundance in the study area at the time of the surveys. This can be estimated as an area-weighted average of the stratum estimates,

$$\begin{aligned} \hat{D} &= \sum_{i=1}^{4} \left(\frac{A_i}{A} \hat{D}_i \right) \\ &= \frac{\hat{E}(s)}{2w\hat{p}} \sum_{i=1}^{4} \left(\frac{A_i}{A} \hat{D}_i \right) \\ &= \frac{\hat{E}(s)M}{2w\hat{p}} \end{aligned}$$

where A_i is the area of stratum i and $A = \sum_{i=1}^{4} A_i$. The corresponding variance, $\hat{\text{var}}(\hat{D})$ can be estimated just as for equation (3) but removing the subscript 1's, and where

$$\hat{\text{var}}(M) = \sum_{i=1}^{4} \left((A_i/A)^2 \, \hat{\text{var}}(M_i) \right)$$

For harbour seals, if the detection function was estimated separately for the water and land components, then the combined density estimate at any given level (by year or PSU within stratum, by stratum or overall) is the mean of the estimates for the two components of the population, and the variance estimate is the mean of the two variance estimates divided by two.

Log-normal, t-based, two-sided 95% confidence limits for the estimates of density and abundance were obtained using equations 3.72-3.76 of Buckland *et al.* (2001).

RESULTS

Realised survey effort

On-effort transects covered in 2004 and 2005 are shown in Figure 1. Table 1 includes area of each stratum and realized survey effort (trackline length). Over 90% of planned trackline effort was realised. Some small segments of trackline were unsurveyed due to poor weather conditions, while others were excluded because they proved in the field to be non-navigable.

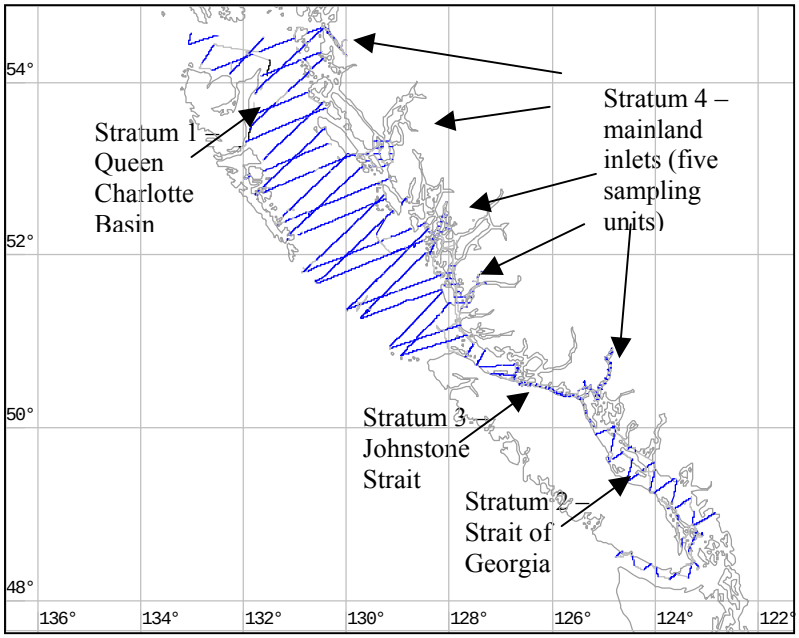

Figure 1. Realised search effort. Tracklines (in blue) followed during the 2004 and 2005 summer field seasons. Only Stratum 1 (Queen Charlotte Basin) was repeated in 2005 (with a new random start point). Note that in most of the area we found almost all the designed lines were navigable (approximately 100% in Strata 1 & 4, and 92% of Stratum 3) and we were able to follow the designed lines. One exception is the US waters south of Vancouver Island, where we could not go for logistical reasons (which resulted in only 64% of the planned tracklines being surveyed). Consequently, the US waters in Stratum 2 were removed from the survey area.

Cetacean sightings

The following summarises the selected detection function for each species. Table 2 lists the truncation distance, number of observations (before and after truncation), fitted detection function model, *p*-value from Kolmogorov-Smirnov (K-S) goodness-of-fit (GOF) test, and estimated mean detection probability ($\hat{p}$) for each species analysed. Table 3 shows the mean cluster size for each species, and Table 4 summarises our species-specific estimates of density and abundance, with corresponding confidence intervals (CIs) and percentage coefficient of variation (%CV, defined as estimated standard error of the estimate over the estimate times 100), by stratum and combined. Locations of each sighting for 12 species are shown in Appendix 1.

Harbour porpoise

A total of 68 harbour porpoise schools were sighted while on-effort (Table 2). The best detection function was found to be a CDS analysis using a hazard rate model with 500m truncation (Figure 2). There was no evidence to support using sightability, cue or aspect as factor covariates, or cluster size as a continuous covariate. None of these analyses was better than the CDS analysis. There was no evidence to support stratification by designed stratum, so a pooled detection function was used.

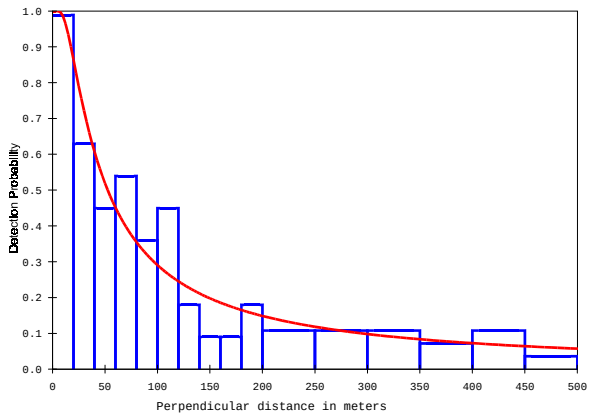

Figure 2. The selected detection function for harbour porpoise sightings.

Dall's porpoise

A total of 112 schools of Dall's porpoise were recorded during the surveys. At 700m truncation, the best model was half-normal with no adjustments (Figure 3) and no covariates.

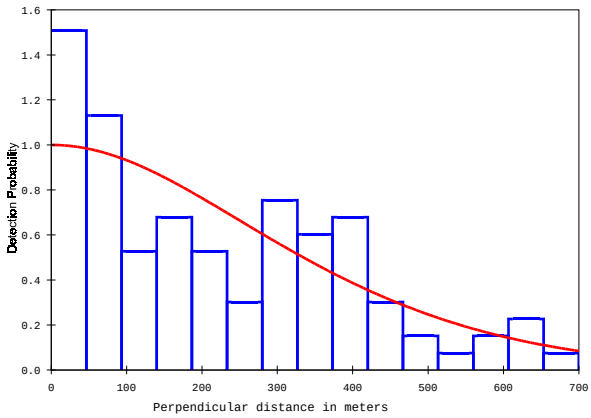

Figure 3. The selected detection function for Dall's porpoise.

Pacific white-sided dolphin

A total of 117 schools of Pacific white-sided dolphins were seen during the survey. These data have a problem with a spike (Figure 4). The selected model was a half-normal detection function truncated at 700m. Note that this was not the model with the lowest AIC (which was the hazard rate model), but the half-normal model was used to avoid fitting the spike at zero distance, which was believed to be an artefact of responsive movement towards the boat (see Discussion). The half-normal was chosen because we felt that the detection function should be qualitatively similar to that for Dall's porpoise (Fig. 3), so we forced the same model (half-normal). The estimated p turned out to be very close to that of Dall's porpoise (Table 2).

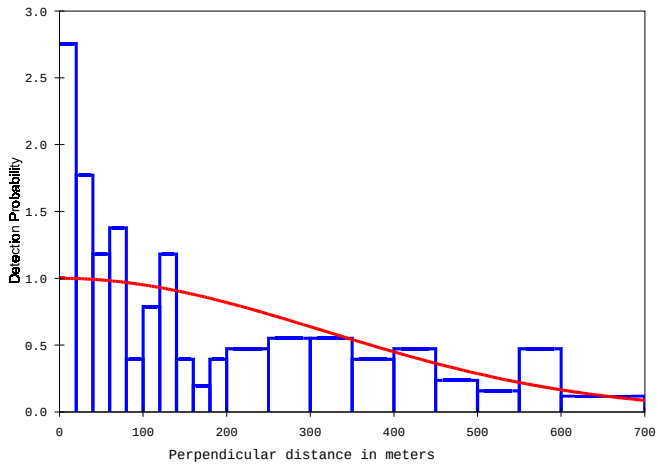

Figure 4. The detection function for Pacific white-sided dolphin sightings.

Humpback whale

A total of 76 humpback whale schools were seen during the study. The model that fitted these data best was a half-normal detection function with one cosine adjustment term, using a 2000m truncation distance (Figure 5).

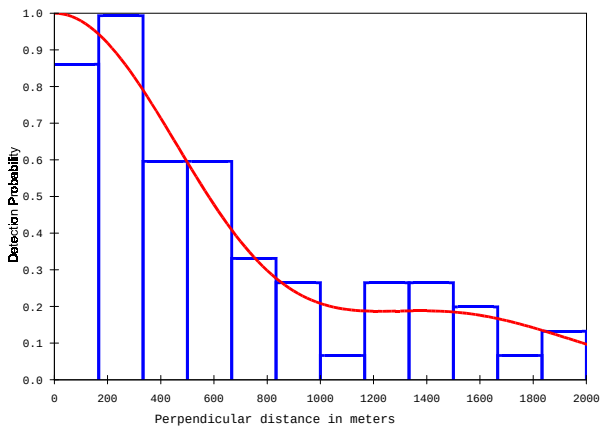

Figure 5. The selected detection function for humpback whales.

Fin whale

A total of 35 fin whale schools were recorded during the study (including three transect-leg sightings used for estimating effective strip width and mean school size, but not for estimating density), which is a smaller sample size than the minimum 60-80 sightings recommended for fitting a detection function (Buckland et al. 2001). However, there was evidence of a shoulder in the detection function (Figure 6), and after truncation at 2000m, the data were described well by a uniform model with one cosine adjustment term.

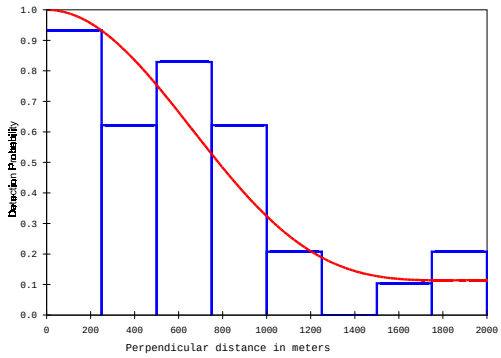

Figure 6. The selected fin whale detection function.

Killer whale

Only 18 killer whale schools were recorded during the survey (15 "residents" and 3 "transients"). Four of these sightings were recorded during transect-leg surveys, and used for estimating effective strip width and mean school size, but not for estimating density. A truncation distance of 1500m was chosen, in that it only required dropping two observations, but provided a reasonable fit to the data (Figure 7) using a half-normal function.

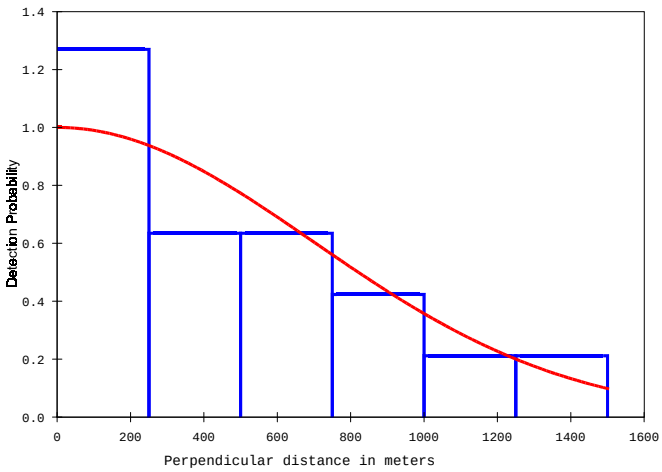

Figure 7. The selected killer whale detection function.

Minke whale

Only 14 minke whale schools were recorded during the survey. We tried fitting a variety of detection functions to the data and found that, with truncation at 300m (that is, truncating one observation and leaving 13), AIC was lowest for a uniform detection function with no adjustments (ΔAIC over uniform + 1 cosine adjustment term was 2.2; and ΔAIC over half-normal was 1.64). In other words, we assumed a strip transect out to 300m (Figure 8). Note that the K-S p-value was p=0.8903 and $\hat{N}$ =388 (CV 26.8%, 95%CI 222-680). For comparison, we tried an analysis with 150m truncation, as the assumption of a strip transect should be even better justified then. While the number of observations was reduced to 8, our results changed little ($\hat{N}$ =475, CV37.3%, 95% CI 221-1,020). So, we conclude that there were fewer than 1,000 minke whales in the area during our study, and most likely around 400 minke whales.

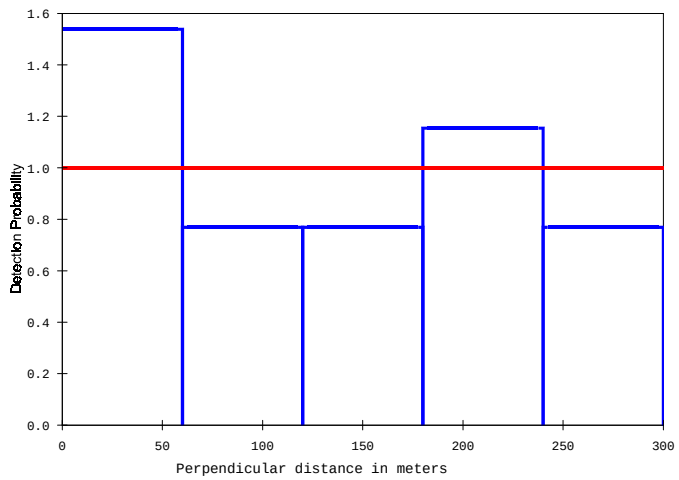

Figure 8. The selected detection function for minke whales.

Pinniped sightings

Harbour seal
There was strong evidence (as indicated by AIC) for stratification our analyses by whether seals were observed hauled out on land, or in water. For each stratum, we used 500m truncation, so that estimated $g(w)$ was >0.1. For both strata, a half-normal detection function was selected (Table 2, Figure 9), however two cosine adjustment terms were preferred for the in-water stratum. Cluster size was estimated separately by stratum as well (Table 3).

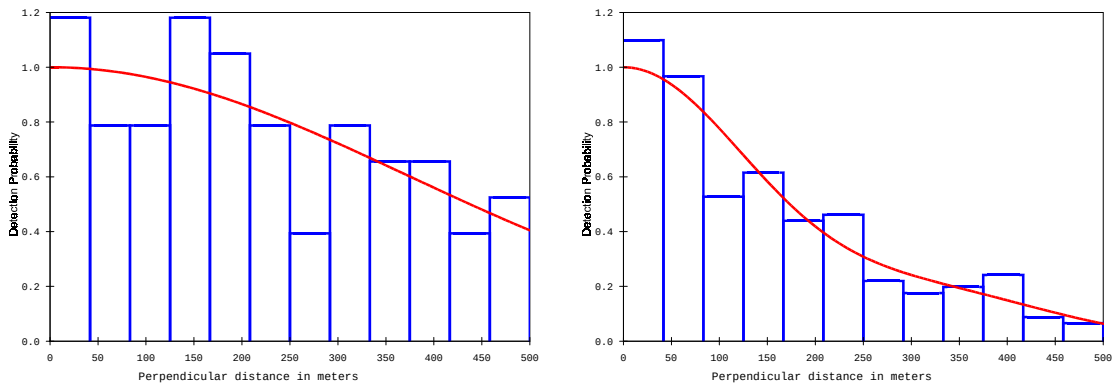

Figure 9. Selected detection functions for harbour seals hauled out (left) and in water (right).

We estimated density and abundance for both parts of the population separately, and combined them by summing density across the parts (see Methods).

Northern elephant seal and Steller sea lion
Too few sightings were made of elephant seals or Steller sea lions to fit a detection function. Locations of sightings are shown in Appendix 1.

DISCUSSION

Preliminary abundance estimates
This first systematic survey of the Inside Passage achieved its primary objectives. Here, we report the first robust estimates of abundance for six cetacean species in the coastal waters of Canada's Pacific region. This information is needed for informing a variety of conservation and management issues, were Canada to assess the sustainability of observed levels of by-catch of

small cetaceans in commercial fisheries (Hall *et al*. 2002), or to pursue proposals to incorporate predator needs when setting fish quotas through ecosystem-based fisheries management (Larkin 1996). In addition, our distribution data provide a much-needed systematic snapshot of the distribution of marine mammals, which we hope will be of use when reviewing permit applications to conduct seismic surveys in the Queen Charlotte Basin region, and if seismic surveys do proceed, for mitigating the impacts of these and other intense anthropogenic noise sources on acoustically sensitive marine mammals. We note that the highest numbers of minke, fin and humpback whales were found in this stratum (Stratum 1), with fin whales found here exclusively.

Our best estimates of abundance throughout the study area (with stratum-specific estimates in Table 4) are: harbour porpoise, 9,120 (95% CI: 4,210-19,760); Dall's porpoise, 4,910 (2,700-8,940); Pacific white-sided dolphin, 25,900 (12,900-52,100); humpback whale, 1,310 (755-2,280); fin whale, 496 (201-1,220); minke whale, 388 (222-680); and northern resident killer whales, 161 (45-574). We estimate that the total abundance of harbour seals in the study area was 19,400 (14,900-25,200). To the best of our knowledge, these are the first abundance estimates for this region for each of these cetacean species except killer whales. While the small sample size makes our northern resident killer whale estimate tenuous, it is worth noting, because this finding can be corroborated against the true number. Our best estimate (161) is very close to the true population size of 219 animals (2004 census, Pacific Biological Station, DFO Canada), and our 95% confidence intervals (45-574) comfortably include this number. While our minke whale abundance estimate is similarly tentative, we are confident in our finding that minke whales were rare. Minke whales were recently deemed to be *Not at Risk* in Canada's Pacific region by the Committee on the Status of Endangered Wildlife in Canada, but our findings indicate that minke whales are just as rare as the *Threatened* fin, humpback and killer whales. The key distinction, of course, is that the minke whale population was not decimated by commercial whaling (Gregr *et al*. 2000) or a live capture fishery (Bigg & Wolman 1975). However, scarcity is a key component of extinction risk (Ehrenfeld 1970). We encourage future research on minke whales to assess the relatedness of BC's inshore minke whales to stocks in Canadian offshore or adjacent US waters.

Also relevant to this study were those species that were not encountered at all – no blue, sperm or right whales were seen, for example. Blue and sperm whales were certainly caught in these waters historically (Gregr *et al*. 2000), although our study area is admittedly on the periphery of preferred habitat for those two species (Gregr & Trites 2001). Right whales do appear in historic catch data for the BC coast, but had already been largely depleted before record keeping began in earnest around 1908 (Gregr *et al*. 2000). Only one sighting of a sei whale was made during our survey (Appendix 1), although sei whales were frequently caught in Queen Charlotte Sound and southern Hecate Strait (our Stratum 1) and this area was predicted to represent good habitat for this species in this region (Gregr & Trites 2001). No beaked whales were seen during our study, despite their tendency to be reported in stranding records with relative frequency along the Queen Charlotte Islands, even though those islands are sparsely populated and do not have a dedicated strandings network (Willis & Baird 1998). Gray whales were also scarce in these inshore waters, but are known to be common on the west coasts of Vancouver Island and Queen Charlotte Islands. We encourage future work to survey the outer coastal waters to complete our assessment of cetacean abundance to the edge of the continental shelf.

Reliability of abundance estimates

The accuracy of our abundance estimates hinges on three primary assumptions: that detection on the trackline was certain; that no responsive movement occurred prior to detection; and that distances and angles were measured without error (Buckland *et al.* 2001, Section 2.1). Measurement error was unlikely to have introduced systematic bias, in that most ranges were measured rather than estimated visually, and visual estimates were calibrated post-hoc using calibration experiments to remove systematic bias. The responsive movement and $g(0)=1$ assumptions remain problematic for some species, and we address the reliability of each species' abundance estimate below. Overall, though, there was no evidence in any of the detection functions to suggest that either of these assumptions introduced major bias in any abundance estimate (except for Pacific white-sided dolphins, see below). However, it will be up to decision-makers and researchers who wish to use our results to decide whether our abundance estimates are sufficient to suit their specific purposes on a case-specific basis.

If it should turn out that these preliminary abundance estimates need to be more precise or accurate – if, for example, by-catch of harbour porpoise were found to be very close to some threshold like potential biological removal (PBR, Wade 1998) that might trigger some management action based on the abundance estimate we report (see Hammond *et al*. 2002, *e.g.*) – then it would be worth investing more time and money to address responsive movement and $g(0)$. (Note that the Canadian government does not use quantitative triggers like a PBR calculation to address by-catch, so this particular example is purely hypothetical.) A similar small-boat survey was conducted recently for coastal dolphins in New Zealand (Dawson *et al*. 2004), where simultaneous helicopter and boat surveys were used to calculate a correction factor for responsive movement and uncertain trackline detection. A similar approach could be used in our region in future. Meanwhile, we began a pilot study in 2005 to conduct double-platform trials to begin to assess how much bias the $g(0)=1$ assumption might introduce, so that we can eventually adjust the abundance estimates reported here accordingly. As expected, the ability to isolate independent platforms on a small boat is poor and consequently our sample size is currently too small to permit statistical analyses, however this did work better on the powerboat than on the sailboat. We will continue to collect double-platform data as opportunities, platform and funding permit, and hope to report on that aspect in future.

Another consideration affecting the reliability of our abundance estimates generally is the need for a large enough sample size of observations for fitting detection function. We certainly need to increase our sample size for minke and killer whales, and preferably for fin whales as well. However, these are all fairly conspicuous animals for which detection functions seemed to have wide shoulders (thereby increasing our statistical power to estimate f(0) reliably). In a similar way, sample size limited our power to assess evidence for bias in estimates because we used pooled detection functions across all strata (and coverage probability was not equal across strata). However, we did try stratifying the detection functions by stratum and found no evidence for doing so.

Over time, one possibility might be to use multiple covariate distance sampling methods (Marques2001) to combine species with detection functions thought to have similar shapes, however our initial investigation showed that single-species models were preferred. More importantly, we have tried MCDS analyses on our existing data. The only analyses where AIC favoured use of MCDS were for Dall's porpoise, for which it make little difference to the estimates, and harbour seals, where we preferred stratification because we suspected that the different sighting processes for seals on land versus in water could have resulted in different shapes for the detection functions. The Dall's porpoise MCDS analysis suggested that our subjective code describing sightability conditions might have improved the detection function fit, but the stratum specific estimates of density using sightability were very similar to the pooled estimate, so it would not have make any difference to our results. Overall, we conclude that if a bias exists due to pooling, we expect it to be small because there was little variability in detectability between strata. We did specifically look at year in Stratum 1 and found no support for including it, even for harbour seals, for which we had a large sample size. However, as our sample sizes and statistical power increase over time, we suspect that MCDS methods may produce slightly better inferences at the stratum level for some species.

A final general comment involves the desire for a "sufficient" number of transects or primary sampling units (PSUs) per stratum. The number of replicate transects was sufficient for Strata 1-3, but we had only 5 PSUs in stratum 4, so our variance estimates there were high (Table 4) and may be unreliable. However, surveying the mainland inlets was a secondary goal, and our overall global variance estimates are quite reasonable (Table 4). Overall assessment by species

Harbour porpoise

Outstanding issues are the apparent spike in the detection function at zero distance (Figure 2) and estimating *g*(0). If the spike at zero distance represented attractive movement, then it would indicate an overestimate of abundance. But harbour porpoise are known to avoid ships (Palka & Hammond 2001), so it may be that this simply reflects the true detection process; namely that we covered a very narrow effective strip width for this species. Future analyses of behavioural notes recorded during the survey will examine whether responsive movement was likely to be a problem, and efforts are underway to increase sample size of double-platform trials to estimate *g*(0). Our current best estimate of abundance of 9,120 animals (95% CI: 4,210-19,760) should be used to assess sustainability of observed levels of by-catch in commercial fisheries. In a simulation framework, one could try a range of plausible values for correction factors for responsive movement and uncertain trackline detection, to assess how robust our management recommendations would be to uncertainty in our abundance estimate.

Dall's porpoise

Our estimate for Dall's porpoise appears to be robust. There is no evidence to suggest that responsive movement was a problem (Figure 3), and *g*(0) should be higher than that for harbour porpoise. Currently, our best estimate of Dall's porpoise abundance in the study area was 4,910 (2,700-8,940).

Pacific white-sided dolphin

Our results for Pacific white-sided dolphins warrant closer attention. The spike near zero in this detection function (Fig. 4) was thought to have arisen due to attractive movement, as this species is known to bow-ride. We corrected for this, by forcing it to fit a half-normal detection function (that is, by not fitting the spike), and so we got a much lower estimated p than the data would otherwise indicate (AIC favoured the hazard rate model). However, if the spike is real (that is, if it arose due to an observation process in which detection probability falls off very sharply), then we have actually solved a problem we did not have, and are now underestimating abundance. And if animals were attracted in from far outside the strip, then we may be overestimating still, as the encounter rate may be higher than it "should be". These issues need to be explored further with future double-platform data collection in which one team searches even farther ahead of the ship than our primary platform was able to do, to assess the point at which responsive movement occurs, and to correct for it (Dawson *et al*. 2004).

We do not expect that the *g*(0) assumption introduced major bias. However, our best estimate of abundance for this species is currently 25,900 (12,900-52,100), and we concur with Heise (1996) that Pacific white-sided dolphins are the most abundant cetacean in BC coastal waters in summer months.

Humpback whale

Our humpback abundance estimate should be robust. The detection function possessed a shoulder, and showed no evidence for responsive movement (Figure 5). Detection probability for large whales should be near 1. There is at present an

international collaboration to use photo-identification to estimate current abundance of humpback whales in the North Pacific basin (http://hawaiihumpbackwhale.noaa.gov/special_offerings/sp_off/splash.html), and we hope that the emerging analyses of that project will give us a new basis for comparison. Our best estimate of humpback whale abundance is 1,310 (755-2,280).

Fin whale

Our humpback abundance estimate should be robust, although our sample size is small. The detection function possessed a wide shoulder, and showed no evidence for responsive movement (Figure 6). Detection probability for fin whales should be near 1. Intriguingly, we saw three times as many fin whale in 2005 as we did in 2004, although given our small sample size, the confidence intervals strongly overlapped. Our best estimate of fin whale abundance is 496 (201-1,220)

Killer whale

Our abundance estimate for killer whales is very preliminary, but in light of our small sample size, the detection function fitted the data surprisingly well (Figure 7). Ideally, we would refrain from estimating abundance with only 18 sightings, but at this rate, one would need two more seasons (which we have no plans to conduct) to have a large enough sample size to begin to assess model fit. However, our relatively imprecise estimate (161) is on the right order of magnitude (true population size in 2004 was 219, Pacific Biological Station, DFO, unpublished data), and our confidence intervals (45-574) comfortably span the true population size. For such a small and highly clustered population though, one would not choose distance sampling *a priori* as the most efficient method to estimate population size. We did go off-effort to collect identification photographs for each killer whale encounter though, and it would be interesting to compare distance sampling to capture-recapture estimates of abundance for this species, given that true population size is actually known.

Minke whale

Our preliminary abundance estimate for minke whales is admittedly tenuous given our small sample size, so it should be interpreted with caution. But while imprecise, there is no reason to expect that the estimate was particularly biased. We decided to fit a strip transect, however responsive movement could have caused animals to enter or leave the strip prior to detection. While responsive movement is not thought to be a problem, a larger sample size will be required to address this quantitatively. Meanwhile, strip transects of 300 and 150m gave roughly the same point estimates of abundance (388 and 475, respectively), with different degrees of precision (as one would expect when sample size varies: 95% CIs 222-680 and 221-1,020, respectively). We have no reason to suspect that detection probability was much lower than 1. Under good conditions and testing a primary platform with only one observer, Williams *et al*. (2006) estimated $g(0)$ for Antarctic minke whales to be approximately 0.90. Our survey was conducted under excellent conditions with a rotating team of 3 observers, so g(0) was likely no lower than 0.9.

Harbour seal

Our abundance estimate for harbour seals is an underestimate due to incomplete trackline detection. Detection of hauled-out seals is not certain, and $g(0)$ is likely to be especially low for those in water. However, as an index of relative abundance, two interesting results emerge. First, our estimates indicate that at on average, roughly two-thirds of the seals were in the water, while one-third were hauled out (Table 4). Secondly, in terms of spatial variability, we report that while the majority of harbour seals were found in the southern straits and in the mainland inlets, a substantial number of animals were in the Queen Charlotte Basin as well (Table 4). Our minimum abundance estimate for BC coastal waters was 19,400 (14,900-25,200).

Future work

Our next steps are to improve the precision and accuracy of our abundance estimates through more data collection and additional analyses, and to begin to apply the distribution data to define areas of important habitat for at-risk species. In terms of additional fieldwork, one obvious need is to increase our sample sizes of observations for minke, fin and killer whales. We hope to address that by re-surveying Stratum 1 in summer 2006, with some replicate surveys in at least one of the primary sampling units in Stratum 4. Secondly, we hope to expand our 2005 pilot study to collect double-platform data to allow us to address outstanding issues of $g(0)<1$ and responsive movement.

On the analysis side, we are investigating two lines of work. The first is to explore applications of improved design-based variance estimators being developed by Buckland and colleagues (R. Fewster and S. Buckland, pers. comm.) for systematic (*c.f.* random) sampling designs. The other area is to apply spatial habitat models of encounter rate, both to reduce variance and also to uncover potential habitat associations (Buckland *et al*. 2004, Hedley *et al*. 1999, Williams *et al*. 2006). Our long-term goal is to build habitat models that help to understand species-specific factors determining marine mammal distribution in our study area. We collected ancillary data (namely water temperature and salinity to a depth of 150m, and zooplankton samples) simultaneously on these surveys, which we plan to use to model factors that influenced cetacean and pinniped distribution and density. We hope to use such results to inform marine planning processes, and to identify candidate areas for protection. Over time, we hope that our replicate surveys will allow us to construct predictive models of animal distribution so that together,

estimates of interannual variability and model uncertainty can inform a quantitative risk assessment framework for exposure of marine mammals to anthropogenic activity.

In spite of the work remaining to be done to improve the estimates from our small-boat surveys though, this report describes the first comprehensive line transect survey of marine mammals in western Canadian waters. We hope that these results form a baseline against which population trends may be measured, and that the results can be of some use in regional, national and international efforts to study, conserve and manage marine mammal populations.

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Table 1. Area of each stratum and realized survey effort

		Area (nm^2)	Number of transects	Total transect length (nm)
Stratum 1	2004	18360	17	963
	2005		18	917
Stratum 2		2422	23	276
Stratum 3		122	24	38.2
Stratum 4	PSU 4	29.2	15	13.0
	PSU 10	164	19	52.3
	PSU 17	94.9	12	25.3
	PSU 21	283	17	60.4
	PSU 29	150	21	38.3
	Total	3489[1]	84[2]	189
Total		24663	166	2383

[1]Total area of stratum 4 is greater than the area of the five primary sampling units that were surveyed

[2]In estimating mean encounter rate for stratum 4, the number of samples are the number of primary sample units (PSUs, i.e. 5), not the total number of transects.

Table 2. Truncation distance, number of observations (before and after truncation), fitted detection function model, *p*-value from Kolmogorov-Smirnov goodness-of-fit test, estimated mean detection probability ($\hat{p}$) for the species analysed.

	w (m)	*n* before	*n* after	model	K-S *p*	$\hat{p}$	%CV($\hat{p}$)
Harbour porpoise	500	68	59	hr	0.99	0.212	32.0
Dall's porpoise	700	112	102	hn	0.19	0.535	7.74
Pacific white-sided dolphin	700	117	98	hn	0.0040	0.551	7.55
Humpback whale	2000	76	70	hn+cos(2)	0.6716	0.386	12.64
Fin whale	2000	35	34[1]	unif+cos(1,2)	0.6780	0.440	18.44
Killer whale (res+trans)	1500	18	16[1]	hn	0.3091	0.564	17.32
Minke whale	300	14	13	unif	0.8903	1.00	0.00
Harbour seal (hauled out)	500	104	70	hn	0.890	0.765	11.5
Harbour seal (in water)	500	246	232	hn+cos(2)	0.515	0.425	7.55

[1]Three fin whale schools and four killer whale schools were sighted during transect-leg (i.e., off-effort) surveys, and were only used in fitting the detection function and estimating mean school size. Three additional killer whale schools that were subsequently identified as transient ecotype were only used in fitting the detection function.

Table 3. Estimated expected cluster size for the species analyzed.

	$\hat{E}(s)$	%CV($\hat{E}(s)$)
Harbour porpoise	1.79	6.24
Dall's porpoise	2.09	6.22
Pacific white-sided dolphin	12.49	17.79
Humpback whale	1.54	6.54
Fin whale	1.56	9.68
Killer whale (residents)	2.38	21.64
Minke whale	1.00	0.00
Harbour seal (hauled out)	3.01	11.5
Harbour seal (in water)	1.13	2.64

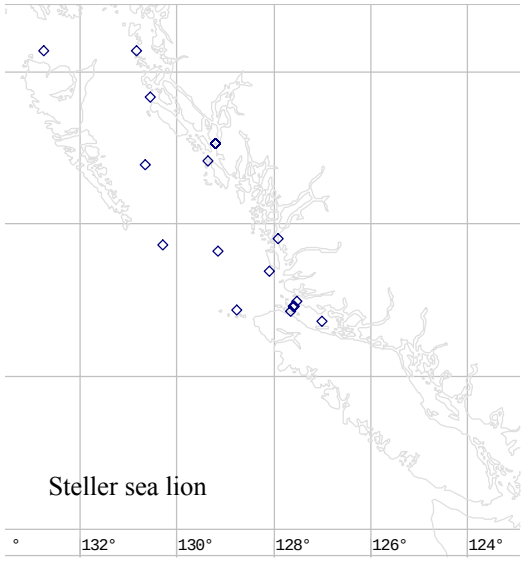
Table 4. Estimated density ($\hat{D}$)and abundance ($\hat{N}$), with corresponding confidence intervals (CIs) and percentage coefficient of variation (%CV), by stratum and combined.

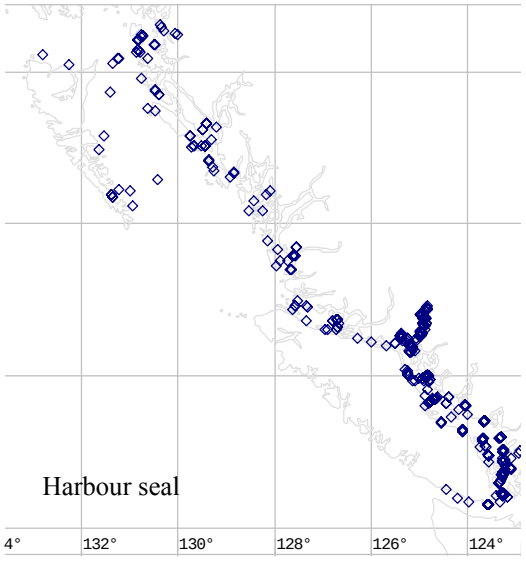
Species	Statistic	Stratum 1			Stratum 2	Stratum 3	Stratum 4	Survey region
		2004	2005	Averaged				
Harbour porpoise	$\hat{D}$	0.179	0.324	0.249	1.421	0	0.327	0.374
	95%CI($\hat{D}$)	0.0413-0.773	0.110-0.960	0.0932-0.670	0.593-3.403	0	0.00828-12.896	0.173-0.811
	$\hat{N}$	3283	5958	4587	3391	0	1140	9120
	95%CI($\hat{N}$)	759-14193	2013-17625	1711-12301	1416-8122	0	29-44989	4208-19764
	%CV	80.5	57.4	51.9	45.9	0	226.0	40.3
Dall's porpoise	$\hat{D}$	0.274	0.198	0.237	0.188	0.407	0.0198	0.202
	95%CI($\hat{D}$)	0.142-0.530	0.0601-0.650	0.123-0.456	0.0757-0.466	0.0829-1.990	0.000483-0.810	0.111-0.367
	$\hat{N}$	5030	3267	4346	448	50	69	4913
	95%CI($\hat{N}$)	2601-9272	1103-11926	2254-8380	181-1113	10-244	1-2825	2700-8938
	%CV	32.4	61.5	32.3	46.3	89.8	223.8	29.2
Pacific white-sided dolphin	$\hat{D}$	1.401	1.243	1.324	0.109	10.98	0	1.064
	95%CI($\hat{D}$)	0.663-2.959	0.395-3.905	0.641-2.734	0.0210-0.562	2.906-41.492	0	0.528-2.140
	$\hat{N}$	25716	22815	24301	260	1344	0	25906
	95%CI($\hat{N}$)	12170-54339	7260-71704	11762-50210	50-1343	365-5081	0	12872-52138
	%CV	37.8	59.6	36.7	94.1	72.2	0	35.3
Humpback whale	$\hat{D}$	0.0577	0.0647	0.0611	0	0	0.0548	0.0540
	95%CI($\hat{D}$)	0.0230-0.145	0.00309-0.135	0.0328-0.114	0	0	0.00222-1.350	0.0310-0.0938
	$\hat{N}$	1059	1187	1122	0	0	191	1313
	95%CI($\hat{N}$)	423-2657	567-2485	601-2093	0	0	8-4709	755-2285
	%CV	46.2	36.9	31.0	0	0	169	27.5

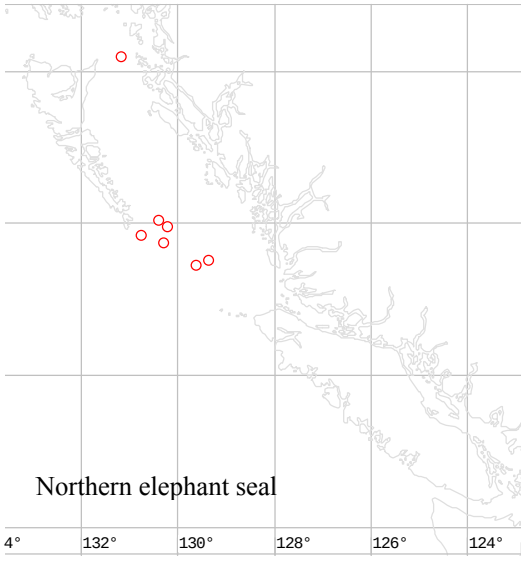
Fin whale	$\hat{D}$	0.0119	0.0429	0.0270	0	0	0	0.0204
	95%CI($\hat{D}$)	0.00469-0.0302	0.0147-0.125	0.0110-0.0663	0	0	0	0.00826-0.0502
	$\hat{N}$	218	787	496	0	0	0	496
	95%CI($\hat{N}$)	86-554	270-2293	202-1218	0	0	0	201-1222
	%CV	47.5	53.3	45.8	0	0	0	45.8
Killer whale (northern resident)	$\hat{D}$	0.00542	0.00855	0.00694	0	0.273	0	0.00661
	95%CI($\hat{D}$)	0.000539-0.0555	0.00191-0.0382	0.00158-0.0306	0	0.0956-0.781	0	0.00185-0.0236
	$\hat{N}$	100	157	128	0	33	0	161
	95%CI($\hat{N}$)	10-1019	35-703	29-562	0	12-96	0	45-574
	%CV	1.54	82.7	81.4	0	55.5	0	67.4
Minke whale	$\hat{D}$	0.0224	0.0168	0.0120	0.0112	0	0	0.0159
	95%CI($\hat{D}$)	0.0117-0.0430	0.00625-0.0453	0.0111-0.0349	0.00180-0.0697	0	0	0.00911-0.0280
	$\hat{N}$	412	309	362	27	0	0	388
	95%CI($\hat{N}$)	215-789	115-832	204-642	44-166	0	0	222-680
	%CV	31.4	49.6	27.7	108.4	0	0	26.8

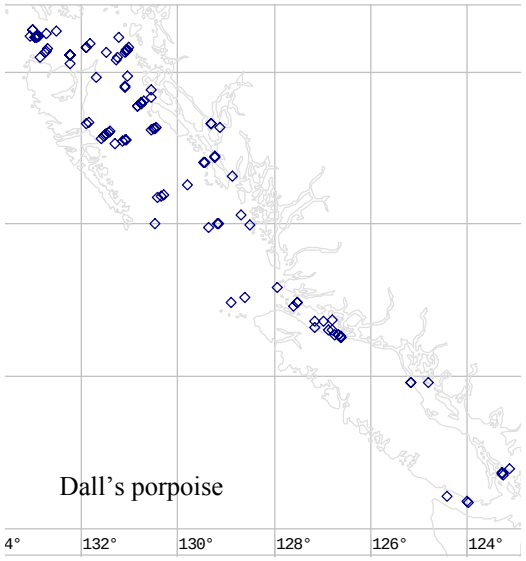
Harbour seal (hauled out)	$\hat{D}$	0.0755	0.0714	0.0735	0.713	0	0.802	0.240
	95%CI($\hat{D}$)	0.0177-0.322	0.0247-0.206	0.0276-0.196	0.225-2.259	0	0.0446-14.5	0.143-0.403
	$\hat{N}$	1387	1311	1350	1702	0	2800	5852
	95%CI($\hat{N}$)	324-5921	455-3777	507-3595	537-5393	0	155-50429	3492-9804
	%CV	78.0	54.2	49.7	60.7	0	1.42	25.9
Harbour seal (in water)	$\hat{D}$	0.148	0.0592	0.105	2.077	0.774	1.88	0.555
	95%CI($\hat{D}$)	0.0710-0.311	0.0242-0.144	0.0575-0.192	1.405-3.069	0.13-26.74	0.360-1.66	0.407-0.758
	$\hat{N}$	2726	1087	1927	4957	95	6545	13524
	95%CI($\hat{N}$)	1302-5705	446-2649	1055-3518	3354-7326	44-204	459-93302	9912-18453
	%CV	36.2	44.4	29.3	19.3	38.5	1.23	15.3
Harbour seal (total)	$\hat{D}$	0.224	0.131	0.178	2.79	0.774	2.678	0.795
	95%CI($\hat{D}$)	0.111-0.452	0.0646-0.264	0.105-0.304	1.83-4.24	0.360-1.66	0.417-17.212	0.612-1.035
	$\hat{N}$	4113	2398	3277	6659	95	9345	19376
	95%CI($\hat{N}$)	2040-8293	1186-4846	1923-5582	4378-10129	44-204	1454-60045	14897-25201
	%CV	35.6	35.9	26.8	21.2	38.5	96.1	13.2

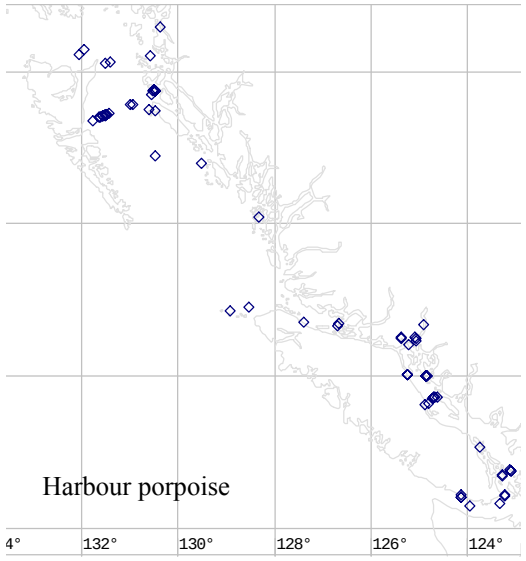
Appendix 1. Sighting locations for 12 marine mammal species.

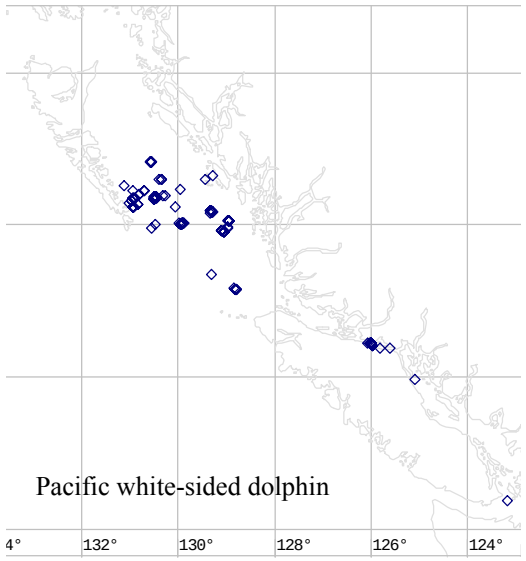


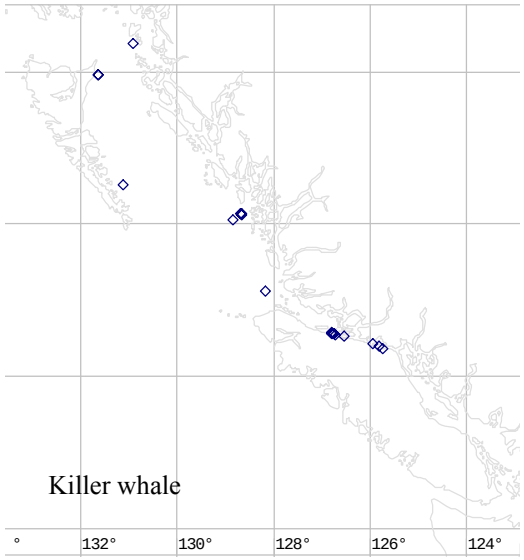


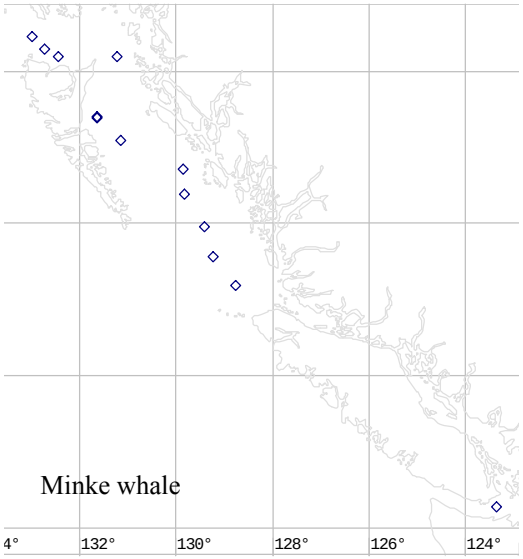


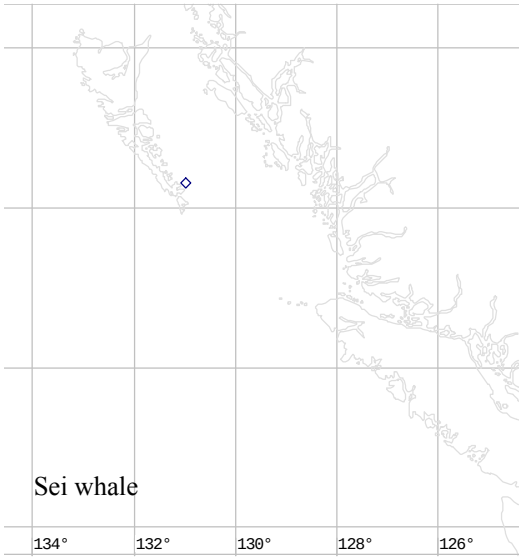


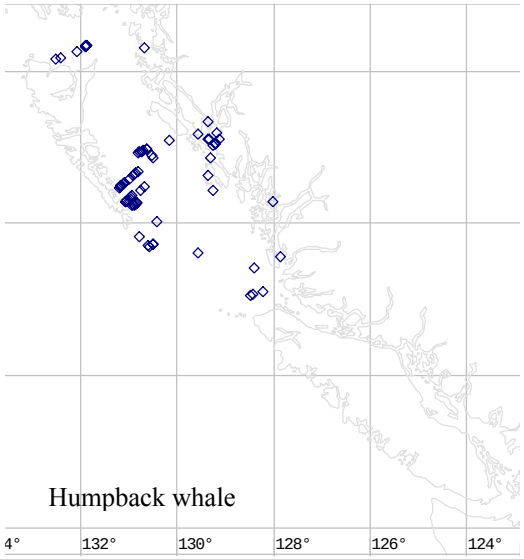





Killer whale
°
132°
130°
128°
126°
124°


Minke whale
4°
132°
130°
128°
126°
124°


Sei whale
134°
132°
130°
128°
126°


Humpback whale
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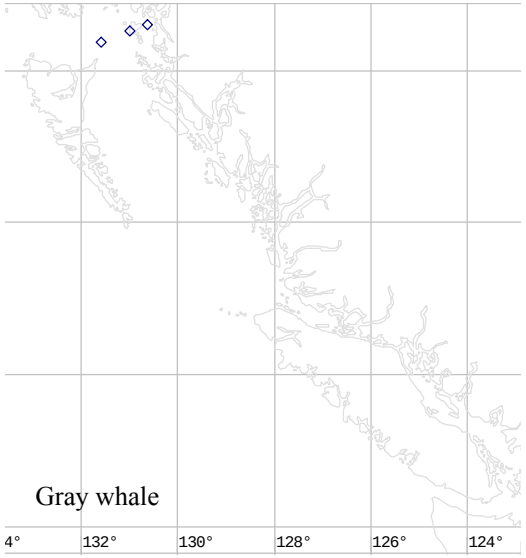
132°
130°
128°
126°
124°


Gray whale
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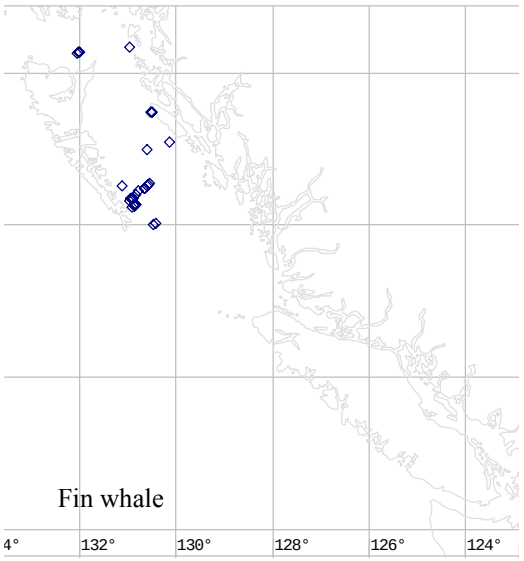
130°
128°
126°
124°


Fin whale
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130°
128°
126°
124°