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 many cetacean species inhabiting the coastal waters of Canada's Pacific coast, including those species that were heavily depleted by commercial whaling. Systematic sightings surveys were conducted in the inshore coastal waters of the Inside Passage, between the British Columbia (BC)-Washington and the BC-Alaska borders. A total of 4,400km (2,400 n.miles) of trackline were surveyed in the summers of 2004 and 2005. Abundance estimates (with 95% confidence intervals) assuming certain trackline detection for seven cetacean species were 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); common minke whale, 388 (222-680); and 'northern resident' killer whale, 161 (45-574). The potential for responsive movement to have affected the accuracy and precision of these estimates is difficult to assess in small-boat surveys. However, the analyses were designed to minimise this factor in the most obvious case (Pacific white-sided dolphins) and pilot data collection has begun to assess the magnitude of the effect and to calculate correction factors for other species. The density of harbour seals, both along the shoreline and at sea, was calculated and it was estimated that total abundance of harbour seals in the study area was at least 19,400 (14,900-25,200). These are new abundance estimates for this region for all cetacean species except killer whales. The small sample size makes the killer whale estimate tenuous, but one worth noting, as it is close to the known number of northern resident killer whales (2004 census was 219 animals, Cetacean Research Program, Pacific Biological Station, Fisheries and Oceans Canada). The common minke whale abundance estimate is similarly tentative, however the results do reveal that common minke whales were relatively rare in this region. 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 provide a systematic snapshot of summertime distribution and abundance of marine mammals in the Queen Charlotte Basin, where offshore oil and gas development and seismic surveys for geophysical research have been proposed to take place. Similarly, the abundance estimates could be used to form the basis of a simulation exercise to assess the sustainability of observed levels of incidental bycatch of small cetaceans in commercial fisheries. The results described here provide a useful reference point to which future survey data can be compared.

KEYWORDS: SURVEY-VESSEL; NORTHEAST PACIFIC; ABUNDANCE ESTIMATE; DISTRIBUTION; HARBOUR PORPOISE; HUMPBACK WHALE; PACIFIC WHITE-SIDED DOLPHIN; MINKE WHALE; DALL'S PORPOISE; KILLER WHALE

INTRODUCTION

Marine mammals in coastal British Columbia

Information on animal distribution and abundance is integral to wildlife conservation and management, however abundance estimates have not been available for many species of cetaceans inhabiting coastal waters of Canada's Pacific coast, including those that were heavily depleted by commercial whaling. More than 20 marine mammal species can be found in the coastal waters of British Columbia (BC), Canada. They 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 the most carefully studied cetacean populations in BC (Bigg *et al*., 1990; Ford *et al*., 2000; Ford *et al*., 1998; Olesiuk *et al*., 1990). Baleen whales were the subject of extensive pelagic and coastal whaling in the northeast Pacific Ocean; the last coastal whaling stations in BC closed in 1967 (Gregr *et al*., 2000). Incidental bycatch of small cetaceans in commercial gillnet fisheries does occur (Hall *et al*., 2002). Whalewatching, once seen as an alternative to whaling, is now considered a potential threat to some cetacean populations via masking effects of boat noise, potential energetic cost of vessel avoidance tactics (Williams *et al*., 2002a; Williams *et al*., 2002b) and disruption of feeding activity (Williams *et al*., 2006) and emission of outboard motor exhaust. All marine mammals in BC, with their 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 existing moratoria 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 (Royal Society of Canada, 2004).

The following is a summary of frequently seen cetacean and pinniped species in BC's inshore waters in summer. 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), have received 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). COSEWIC uses a variety of information sources to assess species' extinction risk (ranging from Extinct, Extirpated, Endangered, Threatened and Special Concern to Data Deficient or Not at Risk) and to report its recommendation to the Canadian government and the

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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

The harbour porpoise (*Phocoena phocoena*) is listed as a species of *Special Concern* in Canada's Pacific waters; a designation indicating that it is considered a species to watch although not in obvious danger of extinction in the near term (COSEWIC, 2003a). Anthropogenic activity, pollution and bycatch have been flagged as conservation threats (Baird, 2003a). Rates of bycatch were estimated by Hall *et al*. (2002) and studies on harbour porpoise habitat usage are taking place off southern Vancouver Island (Hall, 2004).

Dall's porpoise (*Phocoenoides dalli*) is thought to be *Not at Risk* in BC (Jefferson, 1990). It is widely distributed, and commonly seen in deep coastal waters.

The Pacific white-sided dolphin (*Lagenorhynchus obliquidens*) is thought to be *Not at Risk* in BC (Stacey and Baird, 1991). This species returned to inshore waters in BC relatively recently after a decades-long absence (Heise, 1997; Morton, 2000), and has been called the most abundant cetacean in the region (Heise, 1997). Interactions with fisheries are rare locally (Hall *et al*., 2002) and in other areas of the North Pacific (Anon., 2000).

The common minke whale (*Balaenoptera acutorostrata*) status is currently under review by COSEWIC. The species has never been a target of coastal or pelagic whalers in western Canadian waters. There is some evidence that individual common minke whales may be resident to inshore coastal waters of Washington State (Dorsey *et al*., 1990), but the species is relatively poorly studied in BC waters.

The humpback whale (*Megaptera novaeangliae*) is 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 BC waters is extensive, collaborative and ongoing (Cetacean Research Program[1], Fisheries and Oceans Canada (DFO)). While the primary focus of DFO's Cetacean Research Program is on humpback and killer whales, sightings of all cetacean species are recorded during the non-randomised surveys that they have conducted since 2002.

The fin whale (*Balaenoptera physalus*) is 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. Photo-identification studies are beginning on this species in this region (coordinated by DFO).

Two fish-eating (i.e. northern and southern 'resident') populations of killer whales 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*., 2000; Ford *et al*., 1998). 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; Olesiuk *et al*., 1990). 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; Williams and Lusseau, 2006); anthropogenic noise and repeated disturbance (Williams *et al*., 2002b); contaminants (Ross *et al*., 2000); and prey availability (Baird, 2001b).

Sei (*Balaenoptera borealis*), blue (*Balaenoptera musculus*) and North Pacific right whales (*Eubalaena glacialis*) are all listed as *Endangered* in BC, due primarily to historic overexploitation (Gregr *et al*., 2000) which resulted in the small current population size. The gray whale (*Eschrichtius robustus*) is listed as a species of *Special Concern* in BC (COSEWIC, 2004), although the population has recovered since its heavy exploitation in the 19th century. The deep-diving and relatively poorly studied beaked whales are rarely reported in BC waters, although they may be more common than 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 its large and increasing population size (Baird, 2001a; Olesiuk, 1999). Conservation concerns include prey availability and illegal and unreported shooting or mortality incidental to fish farming operations (Baird, 2001a). Their widespread distribution makes them less vulnerable to oil spills than those species that haul out in few locations, although their tolerance of urbanised habitat lends them susceptible to bioaccumulation of contaminants. The 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, 1999).

The Steller sea lion (*Eumetopias jubatus*) is listed as a species of *Special Concern* in western Canada (COSEWIC, 2003c). While the species is locally abundant and the population 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.

The northern elephant seal (*Mirounga angustirostris*) is 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 sightings survey of Canada's Inside Passage

We conducted a line transect survey in the inshore western Canadian waters between the BC-Washington and the BC-Alaska borders. The primary objective of the survey was to generate design-unbiased estimates of abundance of marine mammal species in BC coastal waters during the summer months. A related goal was to provide estimates of at-sea distribution, to begin to understand which areas of the coast

[1] Cetacean Research Program: *http://www.pac.dfo-mpo.gc.ca/sci/sa/cetacean/default_e.htm*

may represent the most important habitat to each species. This paper reports results from our systematic survey of marine mammals of the Inside Passage, which were completed in the summers of 2004 and 2005.

METHODS

Survey design

The survey design is described in detail in Thomas *et al.* (2007). A stratified survey design was used 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. Area names are provided in the map shown in Fig. 1.

(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 82 n.miles offshore in the west to an eastern boundary line drawn down the edges of the outer coastal islands, east of which we considered part of a mainland inlet stratum (see Stratum 4). The southern boundary was the narrow neck of Johnstone Strait.
(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 Geographical Information System (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.

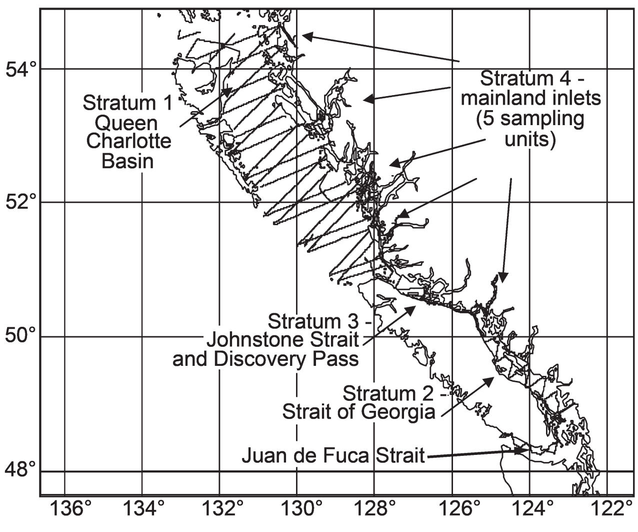

Fig. 1. Realised search effort: tracklines followed during the 2004 and 2005 summer field seasons. Stratum 1 (Queen Charlotte Basin) was re-surveyed in 2005, with the same amount of trackline effort allocated in 2004 but with a new random start point.

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

Field methods to measure animal density

Visual search effort

Data were collected aboard the motor-sailing vessel *Achiever* (a 21m steel-hulled sailboat) in 2004 and the *Gwaii Haanas* (a 20m aluminium power boat) in 2005. An aluminium platform was built for *Achiever* to increase the eye height of observers well above the ship's boom, and both vessels steamed at approximately 8 knots (15km h^{-1}) during searching effort. 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 in both years. 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 used 7×50 or 8×50 binoculars to search 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 group size estimation when needed.

A Global Positioning System (GPS) was connected to a computer running *Logger* software (Logger 2000, International Fund for Animal Welfare). 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 15min, 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. Information collected on factors that could affect sighting conditions included sea-state, cloud cover and precipitation and a subjective sightability code.

As well as cetaceans, pinniped sightings were also recorded, both in the water and hauled out. In searching for hauled out pinnipeds, the shoreline was scanned at the end of each transect carefully, concentrating particularly on the region where the transect line met the shore. For the purposes of analysis, animals seen hauled out on the shore past the end of a transect line were treated as if they were seen right at the end of the line, that is, just within the study area.

Sightings

Whenever a sighting was made of a marine mammal, 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 7×50 reticle binoculars or a graduated perpendicular sighting gauge. Distance to pinnipeds was recorded when possible using a *Bushnell Yardage Pro* laser rangefinder. If a visual estimate had to be made, then those radial distance estimates were corrected subsequently using observer-specific distance estimation experiments. During distance estimation experiments, observers recorded their visual estimates of distance to 20 continuously visible targets (rocks, floating logs, etc.) to which a data recorder measured range using a laser rangefinder. A linear regression model with error proportional to true distance was fitted to the data. Visual estimates from the survey were subsequently corrected by dividing estimated distance by the estimated slope through the origin. No attempt was made to assess whether range or angle *measurements* (i.e. those ranges measured using binocular reticles, rangefinders or a sighting gauge) 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 (swimming normally [travel/forage], avoid, approach, feeding, breaching, other), and its heading relative to the ship (profile, head-on, tail-on or other/unsure).

When time permitted, a decision would be made to 'close' on certain sightings to confirm species identification (usually for balaenopterids); to allow collection of identification photographs of humpback and killer whales; or to obtain more accurate estimates of school size. When the ship left the trackline, search effort was terminated. Search effort was resumed once the ship reached cruising speed and rejoined the original trackline. We also 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

To analyse the data, the methods described by Buckland *et al*. (2001) were followed, which are referred to as 'conventional distance sampling' (CDS) in this paper. Note that these methods assume that probability of detection of animals at zero distance from the trackline is one (the so-called $g(0)=1$ assumption), an issue we return to later. The analysis can be split into three parts: (i) fitting a detection function $g(x)$, where x is perpendicular distance, to observed distances of sightings from the transect to estimate average probability of detection, p; (ii) using observed school sizes to estimate mean school size in the population, $E(s)$; (iii) estimation of animal density, D, using the formula

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

where n is the number of schools seen within w; w is the truncation distance; and L is the total length of the transects searched on effort. 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

Models were fitted to the observed distribution of distances based on the key function and series expansion formulation of Buckland *et al*. (2001). The uniform, half-normal and hazard-rate key functions were used, together with polynomial or cosine series expansion terms as required. The model that minimised the Akaike Information Criterion (AIC) was selected, unless the behavioural observations indicated a problem with avoidance or attraction, in which case the model that we felt best reflected the relationship between probability of detection and distance for that species was subjectively chosen. In practice the lowest AIC model was not chosen in only one case: Pacific white-sided dolphin (see Results). The absolute fit of models was judged using diagnostic plots and the Kolmogorov-Smirnov goodness-of-fit test (Buckland *et al*., 2004).

The analysis of each species' data was begun by considering the need for truncation of the largest distances, since no truncation of sightings was performed in the field. To do this, detection functions were fitted to data with 0, 5 and 10% of the most distant sightings removed and the least amount of truncation necessary was chosen 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 $g(w)>0.10$. The truncation distance was then rounded to the nearest 100m. In general, the amount of truncation used had very little effect on the results.

As the survey design involved stratification, the possibility of fitting a separate detection function in each stratum for species was investigated where sample size was sufficient (>60 observations per stratum). For each of these species, the AIC for a detection functions fit was compared to all data, with the sum of AICs for detection functions fit to data for each stratum separately. The stratified detection functions were used if they had a lower AIC than the pooled function.

Data collected for one pinniped species, the harbour seal were also analysed. It was anticipated that there would be a qualitatively different detection function for seals seen in the water compared with those hauled out, so fitting detection functions stratified by in/out of water was attempted.

Estimation of mean school size

The default method in *Distance* was used to obtain an unbiased estimate of mean school size, as follows. The natural logarithm of school size, $\ln(s)$, was regressed on the estimated probability of detection at the distance the school was seen. The predicted value of $\ln(s)$ at zero distance (where detection probability is 1) was then back-transformed to provide the required estimate.

For harbour seals, it was expected that the school size would be different for observations in the water and seals that were hauled out, so the two groups were analysed separately.

Density, abundance and variance estimation

For each species, density, abundance and associated measures of uncertainty were estimated within each stratum and then combined to produce results for the whole study area. For harbour seals, these statistics were estimated 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 stratum 1, the estimate of mean density for both years was calculated as the effort-weighted mean of the year-specific estimates. For stratum 4, the estimate of mean density was an unweighted mean of the estimates in each PSU, rather than an effort-weighted mean (see Discussion for an alternative). Overall mean density for the study area was calculated as the area-weighted average of the stratum estimates.

Variances were calculated using the delta method, and 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 Fig. 1. Table 1 includes area of each stratum and realised survey effort (trackline length). 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. Nearly 100% of planned survey effort was realised in strata 1 and 3, and 92% of stratum 4. The US waters south of Vancouver Island (stratum 2), were eliminated for logistical reasons, which resulted in only 64% of those planned tracklines being surveyed. Consequently, the US waters in stratum 2 were removed from the survey area at the abundance estimation step.

Table 1

Area of each stratum and realised survey effort.

		Area (n.miles^2)	Number of transects	Total transect length (n.miles)
Stratum 1	2004	18,360	17	963
	2005		18	917
Stratum 2		2,422	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	3,489[1]	84[2]	189
Total		24,663	166	2,383

[1]Total area of stratum 4 is greater than the area of the five primary sampling units (PSUs) that were surveyed.
[2]In estimating mean encounter rate for stratum 4, the number of samples are the number of PSUs (i.e. 5), not the total number of transects.

Cetacean sightings

The following section summarises the number and behaviour of animals sighted during our surveys and some technical issues relating to 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 of observed schools ($\hat{p}$) for each species analysed. Fig. 2 shows the selected detection function for each species. In no cases did detection functions fitted separately to each stratum have a lower AIC than those fit to all data pooled, so the pooled functions were used. Table 3 shows the mean school size for each species, both from the observed data and the size-bias regression, as well as summary statistics on observed group sizes. Table 4 summarises our species-specific estimates of density and abundance, with corresponding 95% confidence intervals (CIs) and percentage coefficient of variation (%CV), by stratum and combined. Sightings distribution maps for 12 of the most frequently seen species are shown in Fig. 3.

Harbour porpoise

A total of 68 harbour porpoise schools was sighted while on-effort (Table 2). Harbour porpoise were seen throughout the study area; most commonly in the southern straits, but also frequently in mainland inlets and Queen Charlotte Basin (Fig. 3). Notes on animal behaviour collected at the time of first sighting do not indicate any severe problem with responsive movement. Most observations were scored as travel/forage (64/68=94%), with the remainder as follows: avoid (3/68=4.4%); and feeding (1/68=1.5%). Data on body aspect relative to the ship showed no obvious signs of responsive movement. Most of the 68 sightings were observed in profile (31 heading left, and 31 heading right), with nearly equal numbers of sightings observed head-on (3) and tail-on (2), and only one sighting of unknown aspect.

The selected truncation distance was 500m, and at this distance the lowest-AIC detection function model was a hazard rate with no adjustment terms (Fig. 2). A half-normal model would have provided a wider shoulder in the estimated detection function, however the AIC for that model was 3.28 higher than the hazard rate. Nevertheless, the half-normal and the hazard rate models resulted in quite similar values for $\hat{p}$ (approximately 25% higher for half-normal: 0.27 vs 0.21).

An outstanding issue for the harbour porpoise sighting data is an apparent spike in the detection function at zero distance (Fig. 2), which might suggest attractive movement or that trackline detection for this species was less than unity (i.e. $g(0)<1$). However, note that little other evidence of attractive movement by harbour porpoise was seen in the data.

Dall's porpoise

A total of 112 schools of Dall's porpoises was recorded during the surveys. Dall's porpoises were seen most commonly in the offshore waters of Queen Charlotte Basin, occasionally in the southern straits, and relatively infrequently in mainland inlets (Fig. 3). Of the 112 sightings, most observations were recorded at the time of first sighting as exhibiting normal, slow-rolling (travel/forage) behaviour (108/112=96.4%), with other behaviours scored as follows: approach (2/112=1.8%); avoid (1/112=0.9%); and feeding (1/112=0.9%). Data on body aspect revealed no obvious evidence of responsive movement. Most of the 112 sightings were observed in profile (48 heading left and 48 heading right), but more observations were scored as head-on sightings (8) than tail-on (5); although 3 were of unknown aspect.

At 700m truncation, the best model was half-normal with no adjustments (Fig. 2), but there was some ambiguity in this case about which detection function best described the observed sightings data. The half-normal, which was used, and the uniform with one cosine adjustment models gave near-identical results in terms of $\hat{p}$ and AIC. There was some support from the data (ΔAIC=1.33) for choosing a hazard-rate model, which would have fitted the apparent spike near zero and resulted in an approximately 50% lower estimate of p (0.35 vs 0.54) and consequently higher estimate of animal density.

While failure of the $g(0)=1$ assumption likely introduced some negative bias in our estimate, obvious cases of Dall's porpoise being attracted to or avoiding our survey vessel were rare (3/112 sightings), which suggests that any bias associated with responsive movement was probably low.

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 ($\hat{p}$) and corresponding percentage coefficient of variation for the species analysed.

	w (m)	n before	n after	Model[2]	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	2,000	76	70	hn+cos(2)	0.6716	0.386	12.64
Fin whale	2,000	35	34[1]	unif+cos(1,2)	0.6780	0.440	18.44
Killer whale (res+trans)	1,500	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. [2]Hr=hazard rate key; hn=half-normal key; unif=uniform key; *cos*(x,y)=cosine adjustment terms of order x and y.

Table 3

Estimated expected school size and corresponding percentage coefficient of variation for the species analysed, plus average, CV (standard error/estimate) and maximum observed school sizes. The minimum school size was 1 for all species.

	Estimated school size		Observed school size			
	$\hat{E}(s)$	% CV	Mean	% CV	Median	Maximum
Harbour porpoise	1.79	6.24	1.81	5.2	2	4
Dall's porpoise	2.09	6.22	2.18	7.5	2	12
Pacific white-sided dolphin	12.49	17.79	18.50	18.4	5	200
Humpback whale	1.54	6.54	1.71	7.2	1	6
Fin whale	1.56	9.68	1.60	13.8	1	8
Killer whale (residents)	2.38	21.64	5.07	28.2	3	25
Minke whale	1.00	0.00	1	0.00	1	1
Harbour seal (hauled out)	3.01	11.5	4.56	11.5	2	35
Harbour seal (in water)	1.13	2.64	1.31	6.3	1	17

Pacific white-sided dolphin

A total of 117 schools of Pacific white-sided dolphins were seen during the survey. Dolphins were most frequently seen in Queen Charlotte Basin, but also in Johnstone Strait and occasionally in the southern straits (Fig. 3). Responsive movement was a bigger problem for this species than with any other. While most observations were recorded at the time of first sighting as exhibiting normal, 'slow-roll' (travel/forage) surfacing behaviour (84/117=71.8%), the second-most frequently recorded behaviour was 'approach' (16/117=13.7%). Remaining behaviours recorded were as follows: feeding (9/117=7.7%); breaching (6/117=5.1%); avoid (1/117=0.9%); and other/unsure (1/117=0.9%). Information on animal heading relative to the ship showed a similar pattern; while most of the sightings were of animals in profile (25 heading left and 30 heading right), there were ten times as many sightings scored as head-on (20) than tail-on (2) and 40 sightings were of uncertain aspect. Not surprisingly, given these observations, the perpendicular distance data exhibit a spike at low distances (note the higher than expected probability of detections at and near 0 perpendicular distance, Fig. 2) that is consistent with attractive movement.

A truncation distance of 700m was chosen, and selected the half-normal detection function with no adjustments. 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 it was felt that the detection function should be qualitatively similar to that for Dall's porpoise (Fig. 2), so the same shape (half-normal) was forced, with the data being used to select the scale (parameter of the half-normal). The estimated p turned out to be very close to that of Dall's porpoise (Table 2).

Humpback whale

A total of 76 humpback whale schools was seen on-effort during the study. Humpback whales were seen most frequently in Queen Charlotte Basin and the mainland inlets of the north and central coasts (Fig. 3). Of the 76 sightings, most observations were recorded at the time of first sighting as exhibiting normal, travel/forage behaviour (66/76=86.8%), with other behaviours scored as follows: breaching (4/76=5.3%); feeding (4/76=5.3%); and other/unsure (2/76=2.6%). No observations were scored as representing avoidance or attractive behaviour, although more animals were observed head-on than tail-on (9 vs 6).

The model that fitted these data best was a half-normal detection function with one cosine adjustment term, using a 2,000m truncation distance (Fig. 2).

Fin whale

A total of 35 fin whale schools was recorded during the study (including three transect-leg sightings used for estimating effective strip width and mean school size, but not for estimating density). All fin whale sightings were made in the Queen Charlotte Basin or adjacent north-coast mainland inlets (Fig. 3). Of the 35 sightings, most observations were recorded at the time of first sighting as exhibiting normal, travel/forage behaviour (30/35=85.7%),

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.

Estimate	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
$\hat{N}$	3,283	5,958	4,587	3,391	0	1,140	9,120
95%CI($\hat{N}$)	759-14,193	2,013-17,625	1,711-12,301	1,416-8,122	0	29-44,989	4,208-19,764
%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
$\hat{N}$	5,030	3,267	4,346	448	50	69	4,913
95%CI($\hat{N}$)	2,601-9,272	1,103-11,926	2,254-8,380	181-1,113	10-244	1-2825	2,700-8,938
%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
$\hat{N}$	25,716	22,815	24,301	260	1,344	0	25,906
95%CI($\hat{N}$)	12,170-54,339	7,260-71,704	11,762-50,210	50-1,343	365-5,081	0	12,872-52,138
%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
$\hat{N}$	1,059	1,187	1,122	0	0	191	1,313
95%CI($\hat{N}$)	423-2,657	567-2,485	601-2,093	0	0	8-4,709	755-2,285
%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	270-2,293	202-1,218	0	0	0	201-1,222
%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($\hat{N}$)	10-1,019	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
$\hat{N}$	1,387	1,311	1,350	1,702	0	2,800	5,852
95%CI($\hat{N}$)	324-5,921	455-3,777	507-3,595	537-5,393	0	155-50,429	3,492-9,804
%CV	78.0	54.2	49.7	60.7	0	1.42	25.9

Cont.

Table 4 cont.

	Stratum 1						
Estimate	2004	2005	Averaged	Stratum 2	Stratum 3	Stratum 4	Survey region
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
$\hat{N}$	2,726	1,087	1,927	4,957	95	6,545	13,524
95%CI($\hat{N}$)	1,302-5,705	446-2,649	1,055-3,518	3,354-7,326	44-204	459-93,302	9,912-18,453
%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
$\hat{N}$	4,113	2,398	3,277	6,659	95	9,345	19,376
95%CI($\hat{N}$)	2,040-8,293	1,186-4,846	1,923-5,582	4,378-10,129	44-204	1,454-60,045	14,897-25,201
%CV	35.6	35.9	26.8	21.2	38.5	96.1	13.2

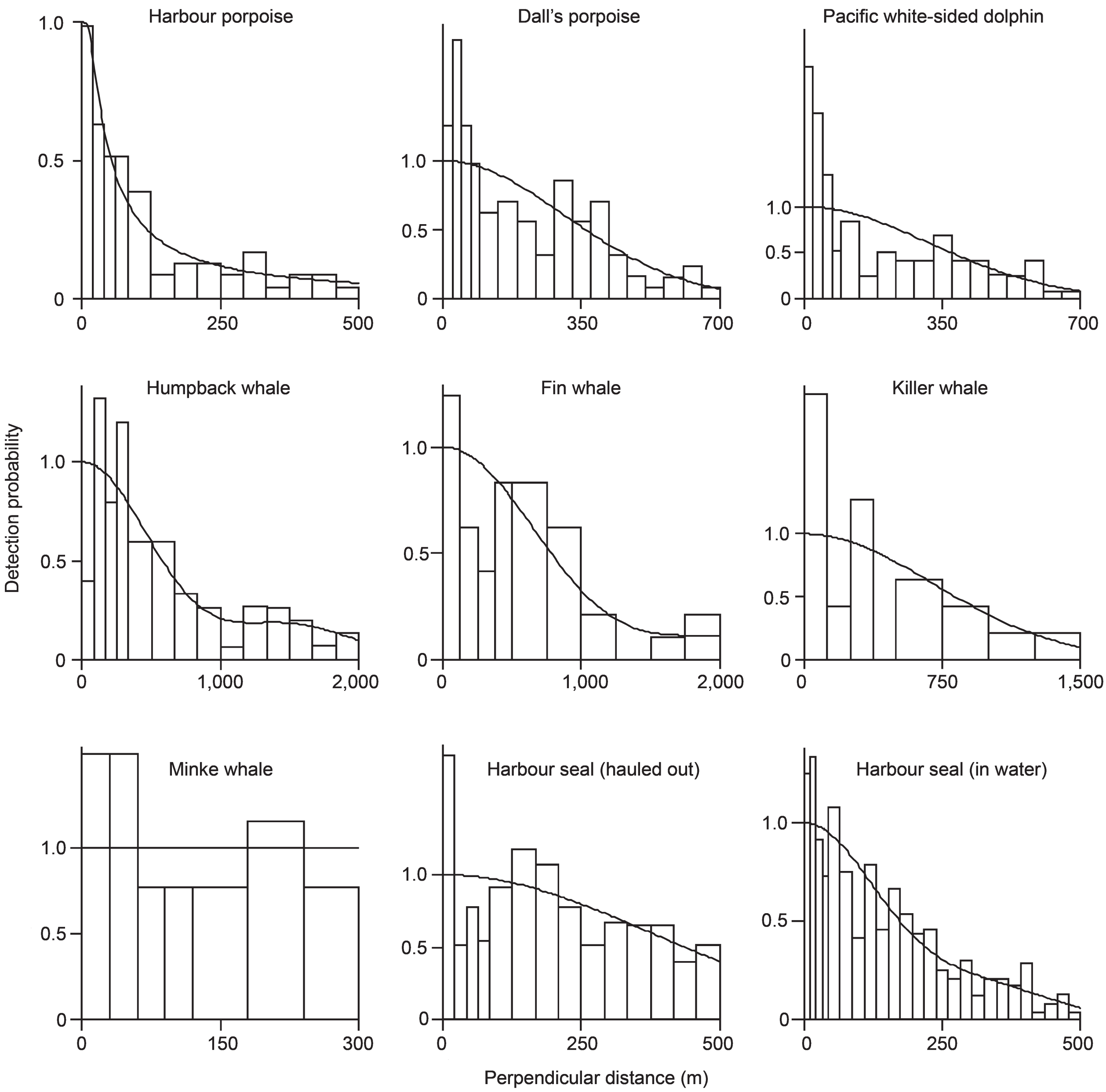

Fig. 2. Histograms of observed distances and fitted detection functions for the species analysed. In constructing the histograms, $1.5\sqrt{n}$ equally spaced intervals were used (where n is the number of detections), except that the first two intervals were further sub-divided into two to make the pattern of detections close to the transect line more clear.

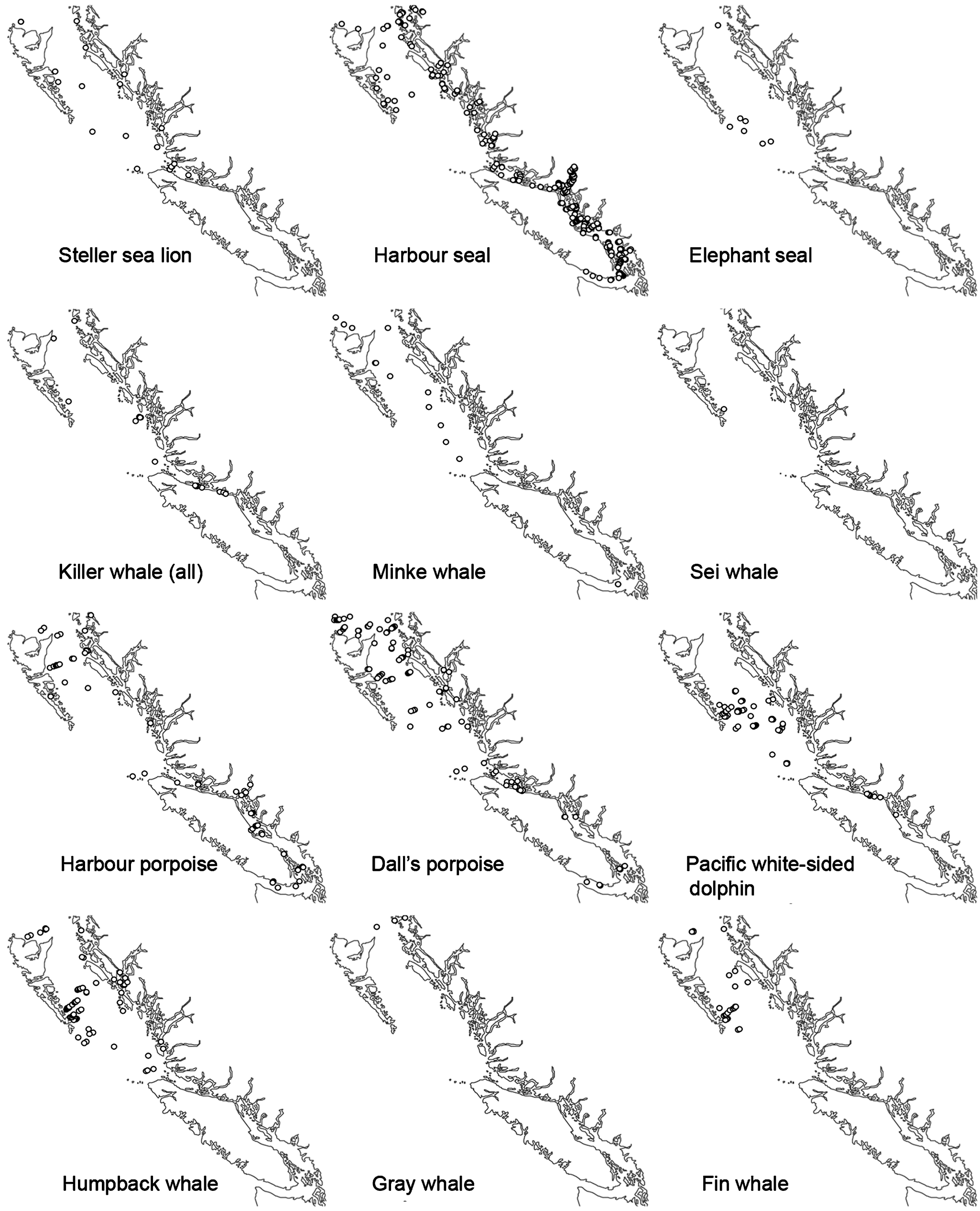

Fig. 3. Maps showing distribution of on-effort sightings, uncorrected for unequal survey effort between strata, for 12 marine mammal species in coastal waters of BC in summer months of 2004 and 2005.

with other behaviours scored as follows: feeding (2/35=5.7%); and other/unsure (3/35=8.6%). No observations were scored as representing avoidance or attractive behaviour. However, while 4 observations were made of animals in tail-on aspect, only 1 head-on sighting was made, as were 8 sightings of unsure aspect.

Despite the small sample size, the perpendicular distance data were reasonably well behaved; there was evidence of a shoulder in the detection function (Fig. 2), and after truncation at 2,000m, the data were described well by a uniform model with one cosine adjustment term.

Killer whale

Only 18 killer whale schools were recorded during the survey (15 'residents' and 3 'transients'). Four of these sightings were recorded during transit-leg surveys and used for estimating effective strip width and mean school size, but not for estimating density. Sightings were most common in Queen Charlotte Basin and Johnstone Strait, and in adjacent north- and central-coast mainland inlets (Fig. 3). No southern resident killer whales were observed while on effort in the southern straits. Most observations of killer whales were recorded as exhibiting normal, travel/forage behaviour (16/18=88.9%), with the remaining two scored as other/unsure (11.1%). No observations were scored as representing avoidance or attractive behaviour. Information on animal heading also failed to show evidence of responsive movement. Equal numbers of head-on and tail-on sightings (2 each) were recorded.

A truncation distance of 1,500m was chosen, in that it only required dropping two observations, but provided a reasonable fit to the data (Fig. 2) using a half-normal function.

Common minke whale

Only 14 common minke whale schools were recorded during the survey. Sightings were distributed more or less uniformly from north to south, but all were seen somewhat offshore; no sightings of minke whales were made in mainland inlets (Fig. 3). Of the 14 sightings, all (14/14=100%) were recorded as displaying normal, travel/forage behaviour at the time of first sighting; no evidence of avoidance or attractive behaviour was observed. Information on animal heading relative to the ship indicated that one animals was observed head-on, and two tail-on.

A variety of detection functions were fitted to the data and it was 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). This model corresponds to a strip transect out to 300m (Fig. 2). Note that the K-S p-value was p=0.8903 and $\hat{N}$=388 (CV 26.8%, 95%CI 222-680). For comparison, an analysis with 150m truncation was done, as the assumption of a strip transect should be even better justified then. While the number of observations was reduced to 8, the results changed little ($\hat{N}$=475, CV37.3%, 95% CI 221-1,020). So, it can be concluded that there were fewer than 1,000 minke whales in the area during the study, and most likely around 400 minke whales.

Pinniped sightings

Harbour seal

Harbour seals were by far the most frequently sighted marine mammal species during the study (Table 2; 350 sightings [104 hauled out, 246 in water]). Sightings were most frequent in southern straits and mainland inlets, but nevertheless quite common in Queen Charlotte Basin. There was strong support from the data (as indicated by AIC) for stratification of detection function estimation by whether seals were observed hauled out, or in water. For each stratum, 500m truncation was used, so that estimated $g(w)$ was >0.1. For both strata, a half-normal detection function was selected (Table 2, Fig. 2); however two cosine adjustment terms were preferred for the in-water stratum. School size was estimated separately by stratum as well (Table 3).

Density and abundance were estimated for each part of the population separately and combined by summing density across the parts (see Methods). The abundance estimate for harbour seals is an underestimate due to incomplete trackline detection. Trackline detection of hauled-out seals is also uncertain, but $g(0)$ is likely to be especially low for those in water. However, as an index of relative abundance, two results emerge. First, the point estimates indicate that at on average, at least two-thirds of the seals were in the water (perhaps much more than this, if $g(0)$ for animals in the water is particularly low), while one-third were hauled out (Table 4). Secondly, in terms of spatial variability, 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).

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 Fig. 3. All sightings of northern elephant seals (7/7) were made in Queen Charlotte Basin, as were the majority (17/21) of Steller sea lions (Fig. 3). Remaining sightings of Steller sea lions (4/21) were made in mainland inlets (Stratum 4). All sightings of elephant seals were of animals at sea (that is, no sightings were made of hauled-out elephant seals), which is unsurprising given that the survey was conducted in a feeding area. Similarly, most Steller sea lion sightings (16/21) were of animals in the water, with the remaining 5/21 sightings hauled out on land.

DISCUSSION

Preliminary estimates of abundance and distribution

This systematic survey of Inside Passage waters achieved its primary objectives and preliminary (see the section on reliability below) estimates of abundance for six cetacean species in the inshore coastal waters of Canada's Pacific region are reported. This information is needed for informing a variety of conservation and management issues, were Canada to assess the sustainability of observed levels of bycatch 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, the distribution data provide a systematic snapshot of marine mammal distribution, which it is hoped 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 animals. It should be noted that the highest numbers of common minke, fin and humpback whales were observed in this stratum (Stratum 1), with fin whales found here exclusively. It should be noted that although the term 'population' is used for convenience in this discussion, this does not imply that the animals found in the surveyed areas necessarily represent discrete biological populations. Rather, the numbers presented represent the estimated numbers of animals found in the surveyed waters at the time of the surveys.

The best estimates of abundance throughout the study area, and the stratum-specific estimates, are presented in Table 4. While small sample size makes the northern resident killer whale estimate tenuous, it is one worth noting, because this is the only finding that can be corroborated against the true number. The best estimate (161) is close to the true population size of 219 animals (2004 census, Cetacean Research Program, Pacific Biological Station, DFO Canada), and the 95% confidence interval (45-574) comfortably includes this number. While the common minke whale abundance estimate is similarly tentative, we are confident in our finding that minke whales were relatively rare. Future research on common minke whales should be encouraged 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 or northern fur seals were seen, for example. Blue and sperm whales were certainly caught in these waters historically (Gregr *et al*., 2000), although the study area was admittedly on the periphery of the preferred habitat for those two species (Gregr and 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 the survey (Fig. 3), although sei whales were frequently caught in Queen Charlotte Sound and southern Hecate Strait (our Stratum 1) and this area was predicted to represent a good habitat for this species in the region (Gregr and Trites, 2001). No beaked whales were seen during the study, despite their tendency to be reported in stranding records relatively frequently along the Queen Charlotte Islands, which are sparsely populated and do not have a dedicated strandings network (Willis and 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. It is hoped to expand the study area to survey the outer coastal waters to complete the assessment of cetacean abundance at least to the edge of the continental shelf. There is at present an international collaboration[2] to use photo-identification to estimate current abundance of humpback whales in the North Pacific basin (SPLASH), and it is hoped that the emerging analyses of that project will give a new basis for comparison within the study region of overlap.

Reliability of abundance estimates

The accuracy of the 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 in radial ranges was unlikely to have introduced major bias, in that most ranges were measured rather than estimated visually, and estimates were calibrated *post-hoc* using calibration experiments to remove systematic bias. The two remaining assumptions are problematic for some species. Summarised below are the major conclusions regarding responsive movement and uncertain trackline detection from the behavioural and animal heading data, the shape of the fitted detection function, and comparisons with other studies. After summarising these outstanding technical issues, the utility of these preliminary results are discussed along with planned future work for conservation and management.

Responsive movement and uncertain trackline detection are outstanding issues for three species: harbour and Dall's porpoises and Pacific white-sided dolphins. For harbour porpoises, the shape of the detection function illustrates outstanding technical issues for this species. If the observed spike in detections near zero distance represented attractive movement, then the selected hazard-rate detection function would result in an overestimate of abundance. However, (1) no evidence was seen of attractive movement in the study; (2) harbour porpoise are generally thought to avoid ships (Palka and Hammond, 2001); and (3) only 4% of the sightings suggested avoidance behaviour. It may be that the observed spike simply reflects the true detection process: that a very narrow effective strip width was covered for this species, possibly because observers were searching with low-power binoculars from a relatively low platform (~5m). A second alternative is that the detection function lacked a wide shoulder because $g(0)$ was substantially less than 1. Efforts are underway to increase sample size of double-platform trials to estimate $g(0)$ for this platform. However, until $g(0)$ can be estimated directly for the survey, it may be useful to consider a range of likely values. Barlow (1995) estimated $g(0)$ due to perception bias for cryptic species (harbour and Dall's porpoises and pygmy sperm whale) to be 0.78 for a team of three people. Barlow (1995) suggested that this may be an overestimate of $g(0)$ for the harbour porpoise, because it is pooled with the more detectable Dall's porpoise. Barlow *et al*. (1997) reviewed $g(0)$ estimates reported from shipboard surveys for harbour porpoises in US waters that ranged from 0.4 to 0.78, depending, *inter alia*, on the number of observers and sighting conditions. Palka (2000) conducted ship-board sightings surveys for harbour porpoises in the Gulf of Maine and Bay of Fundy that produced estimates of $g(0)$ ranging from 0.25 to 0.74, depending on platform height (9 or 14m above the sea surface), number of observers and stratum. Clearly, our estimate of harbour porpoise abundance is negatively biased to some degree.

For the Dall's porpoise, responsive movement has been found to be a large problem in some studies (Buckland and Turnock, 1992), and not in others (Barlow, 1995). The behavioural data presented here suggest that Dall's porpoise were detected before most animals started to respond to the ship, which may reflect the tendency for observers to search as instructed, namely ahead of the ship using binoculars, or suggest that the small, relatively quiet research vessel elicited weaker avoidance responses from Dall's porpoise than has been reported from larger ships. The assumption that $g(0)=1$ no doubt did introduce some negative bias into the abundance estimate, but trackline detection probability may be higher for Dall's porpoise than for the more cryptic harbour porpoise. Other studies have reported estimates of $g(0)$ for Dall's porpoise of 0.78 (Barlow, 1995) and 0.6 (Buckland and Turnock, 1992).

For the Pacific white-sided dolphin, the observed spike in detections near zero distance (Fig. 2) was thought to have arisen due to attractive movement, as this species is known to bow-ride; the behavioural and animal heading data indicated a responsive movement problem. An attempt to correct for this was done by fitting a half-normal detection function to the data (that is, by not fitting the spike). As a result a much lower estimated p was obtained than the data would otherwise indicate (because AIC favoured the hazard rate model). However, if the spike occurred due to a sharp decline in detection probability with increasing distance, rather than responsive movement, then the decision not to fit the spike means that abundance would be underestimated. Alternatively, if animals were attracted in from far outside the surveyed strip, then overestimation may have occurred, as the encounter rate will be higher than it would be if there were no responsive movement. These issues need to be explored further with future double-platform data collection in which one team searches farther ahead of the ship than the primary platform was able to do, to assess the point at which responsive movement occurs, and to correct for it (Dawson *et al*., 2004; Palka and Hammond, 2001). In contrast to the responsive movement problem, it is not expected that the $g(0)=1$ assumption introduced major bias to the dolphin abundance estimate. Barlow (1995) reported an estimate of $g(0)$ for large delphinids of 0.736 for small groups (1-20 animals) and 1.0 for large (>20) groups; the mean observed school size was approximately 19 animals (Table 3). The best estimate of abundance for this species is the highest estimated for any cetacean species in the area, and we concur with Heise (1997) that Pacific white-sided dolphins are the most abundant cetacean in BC coastal waters in summer months.

In contrast to the small cetacean species, the abundance estimates for whale species seem robust to the responsive movement and $g(0)=1$ assumptions. The humpback whale abundance estimate should be reasonably robust. The detection function possessed a shoulder and responsive

[2] *http://hawaiihumpbackwhale.noaa.gov/special_offerings/sp_off/splash.html*

movement was not found to be a problem. Negative bias due to $g(0)<1$ was unlikely to have been large. Estimates of $g(0)$ for large whales due to perception bias have ranged from 0.9-1.0 (Barlow, 1995; Williams *et al*., 2006) under calm conditions. Detection probability for fin whales should be near 1 (Barlow, 1995; Williams *et al*., 2006). For fin whales, the primary issue here is simply one of small sample size.

The abundance estimate for killer whales is very preliminary, but in light of the small sample size, the detection function fitted the data surprisingly well (Fig. 2). Ideally, abundance would have been estimated with more than 18 sightings, but at this rate, one would need two more seasons to have a large enough sample size to begin to assess model fit. However, this relatively imprecise estimate (161) is of the right order of magnitude (true population size in 2004 was 219, Cetacean Research Program, Pacific Biological Station, DFO), and the confidence interval (45-574) comfortably spans 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. Since identification photographs for each killer whale encounter were collected off-effort, it is possible to compare distance sampling to capture-recapture estimates of abundance for this species, in addition to the true, known population size.

For common minke whales, the preliminary abundance estimate is admittedly tenuous given the small sample size and should be interpreted with caution. But at this stage, increasing the precision of the estimate for this species is of greater concern than for reducing bias. A strip transect was fitted, 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. Meanwhile, strip transects of 300 and 150m gave roughly the same point estimates of abundance (388 and 475, respectively), with different precision (as one would expect when sample size varies: 95% CIs 222-680 and 221-1,020, respectively). Detection probability was certainly less than 1 for minke whales. Perception bias may not have introduced much negative bias in the abundance estimate. Under excellent sighting conditions, Williams *et al*. (2006) estimated $g(0)$ for Antarctic minke whales to be approximately 0.9 due to perception bias and Barlow (1995) has reported an estimate of $g(0)$ due to perception bias for small whales of 0.840. However, attempts to address availability bias have produced substantially lower estimates of $g(0)$ for minke whales. Skaug and Schweder (1999) have estimated that 56 to 68% of minke whales on the trackline may be missed by observers during North Atlantic surveys.

The pinniped abundance estimates will require increased sample size in the case of Steller sea lions and elephant seals and estimates of $g(0)$ for all three species.

Nevertheless, these preliminary estimates are a good starting point for discussion, which in turn will help to prioritise future research activities. If it should turn out that precise, accurate estimates of absolute abundance are needed for management purposes – if, for example, bycatch of harbour porpoise were found to be near some threshold like Potential Biological Removal (Wade, 1998) that might trigger some management action based on the abundance estimate reported (e.g. Hammond *et al*., 2002) – then it would be worth investing more time and money to address responsive movement and $g(0)$. 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 BC in future. A pilot study was initiated in 2005 to conduct double-platform trials to begin to assess how much bias the $g(0)=1$ assumption might introduce, so that the abundance estimates reported here can be adjusted accordingly. As expected, this attempt to isolate independent platforms on a small boat was problematic and consequently the sample size is currently too small to permit statistical analyses; however this did work better on the powerboat (2005) than on the sailboat (2004). Double-platform data will continue to be collected as opportunities, platform and funding permit, and we hope to report on this in future. In the meantime, we plan to explore the sensitivity of PBR calculations to varying levels of bias and precision in our porpoise abundance estimates in a simulation framework.

Another consideration affecting the reliability of the abundance estimates is the need for a large enough sample size of observations for fitting the detection function. This may be a particularly important issue for the estimates of abundance for small cetaceans. The decision to use AIC to guide the choice of detection function (unless there was good reason not to do so) seems sensible, but requires closer inspection. For Dall's porpoise, the two detection functions with lowest AIC (half-normal and uniform) gave similar results, but one with only marginally higher AIC (hazard rate) would have resulted in a much higher estimate of animal density. For Pacific white-sided dolphins, the behavioural data gave strong support for not choosing the detection function with the lowest AIC, that is, for not fitting the spike near zero distance, but it is unable to determine whether the analyses addressed responsive movement issues completely. For that, work on better double-platform data collection or the use of higher power binoculars are required. For harbour porpoise, the detection function lacked a shoulder, but the next-best detection function produced a much worse AIC and gave similar results in any case.

Future work

The next steps are to improve the precision and accuracy of the abundance estimates through more data collection (funding permitting) and additional analyses and simulations, 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 sample sizes of observations for common minke, fin and killer whales. Secondly, it is hoped that the 2005 pilot study can be expanded to collect double-platform data, to be used to address outstanding issues of $g(0)<1$ and responsive movement.

Over time, one possibility might be to expand the analyses to use multiple covariate distance sampling methods (MCDS); (Marques and Buckland, 2003; Marques, 2001), in which models allow factors (such as group size or sighting conditions) to alter the scale of the detection function without affecting its shape. MCDS offers a parsimonious intermediate between full stratum-level stratification of the detection function and full pooling. Note that covariates were tried and found that no MCDS model was selected over CDS, with two exceptions. The only analyses where AIC favoured the use of MCDS were for Dall's porpoise, for which it made little difference to the estimates, and for harbour seals, where stratification was preferred because it was suspected that the different sighting processes for seals on land versus in water could have resulted in different shapes for the detection functions. More specifically, the Dall's porpoise MCDS analysis suggested that the 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 the 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. Year was specifically looked at as a covariate in stratum 1 and no support for including it was found, even for harbour seals, for which there was a large sample size.

A more salient application of MCDS methods might be to improve the estimates for species with very small sample sizes, by using MCDS to model detection function for multiple species simultaneously. It is possible to combine species with detection functions thought to have similar shapes, however the initial investigation showed that single-species models were preferred. As our sample sizes and statistical power increase over time, it is suspected 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 there were only 5 PSUs in stratum 4, so the variance estimates there were high (Table 4) and may be unreliable. However, surveying the mainland inlets was a secondary goal, and the overall global variance estimates are quite reasonable (Table 4).

Two additional lines of work are being investigated in terms of reanalysis of existing data. 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 (e.g. Hedley *et al*., 1999; Williams *et al*., 2006). The long-term goal is to build habitat models that help to understand species-specific factors determining marine mammal distribution in our study area. Ancillary data (namely water temperature and salinity to a depth of 150m, and zooplankton samples) were collected simultaneously on these surveys, which will be used to model factors that influence cetacean and pinniped distribution and density. Such results could be used to inform marine planning processes and to identify candidate areas for protection. Over time, it is hoped that the replicate surveys will allow predictive models of animal distribution to be constructed 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 this small-boat survey, this report provides the first comprehensive line transect survey of marine mammals in these waters. Such surveys, undertaken with a limited budget, can still be valuable when carefully designed (Thomas *et al*., 2007), planned and implemented, as was the case in this low-cost university-NGO partnership. We hope that these results form a baseline against which population trends may be measured in future, and that in the meantime, these preliminary results can be of use in regional, national and international efforts to study, conserve and manage marine mammal populations.

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