Potential limits to anthropogenic mortality of small cetaceans in coastal waters of British Columbia

Rob Williams, Anna Hall, and Arliss Winship

Abstract: Small cetaceans are by-caught in salmon gillnet fisheries in British Columbia (BC) waters. In Canada, there is currently no generic calculation to identify when management action is necessary to reduce cetacean bycatch below sustainable limits. We estimated potential anthropogenic mortality limits for harbour (*Phocoena phocoena*) and Dall's (*Phocoenoides dalli*) porpoises and Pacific white-sided dolphins (*Lagenorhynchus obliquidens*) using quantitative objectives from two well-established frameworks for conservation and management (the United States' Marine Mammal Protection Act and the Agreement on the Conservation of Small Cetaceans of the Baltic and North Seas), which are similar to some management objectives developed for marine mammal stocks elsewhere in Canada. Limits were calculated as functions of (*i*) a minimum abundance estimate (2004–2005); (*ii*) maximum rate of population increase; and (*iii*) uncertainty factors to account for bias in abundance estimates and uncertainty in mortality estimates. Best estimates of bycatch mortality in 2004 and 2005 exceeded only the most precautionary limits and only for porpoise species. Future research priority should be given to determining small cetacean stock structure in BC and refining species-specific entanglement rates in these and other fisheries. The approach offers a quantitative framework for Canada to meet its stated objectives to maintain favourable conservation status of cetacean populations.

Résumé : Des petits cétacés se retrouvent parmi les captures accessoires des pêches au saumons au filet maillant dans les eaux de la Colombie-Britannique (BC). Au Canada, il n'y a pas de calcul générique pour déterminer quand des actions de gestion sont nécessaires pour réduire les prises accessoires de cétacés sous un seuil durable. Nous estimons les limites potentielles de la mortalité due aux facteurs anthropiques chez le marsouin commun (*Phocoena phocoena*), le marsouin de Dall (*Phocoenoides dalli*) et le lagénorhynque à flancs blancs du Pacifique (*Lagenorhynchus obliquidens*) à l'aide d'objectifs quantitatifs provenant de deux cadres de référence bien établis pour la conservation et la gestion (la loi américaine sur la protection des mammifères marins et l'accord sur la conservation des petits cétacés dans la Baltique et la mer du Nord), qui possèdent certains des objectifs mis en place pour les stocks de mammifères marins ailleurs au Canada. Les limites ont été calculées en fonction (*i*) d'une estimation d'abondance minimale (2004–2005), (*ii*) du taux maximal d'accroissement de la population et (*iii*) de facteurs d'incertitude qui tiennent compte de l'erreur dans les estimations d'abondance et de l'incertitude dans les estimations de mortalité. Les meilleures estimations de la mortalité due aux captures accessoires en 2004 et 2005 ne dépassent que les limites les plus prudentes et seulement dans le cas des espèces de marsouins. Les priorités futures de recherche devraient inclure la détermination de la structure du stock des petits cétacés en BC et la précision des taux d'empêtrement en fonction des espèces dans ces pêches et dans les autres. Notre approche fournit un cadre quantitatif pour que le Canada atteigne ses objectifs avoués de maintenir un état de conservation favorable pour les populations de cétacés.

[Traduit par la Rédaction]

Introduction

The effects of fishery-related mortality on small cetacean populations around the world have long been known to be an important conservation factor (Read et al. 2006). For some, such as vaquita (*Phocoena sinus*), it may be the driving factor in population decline (D'agrosa et al. 2000). In 1991, the International Union for the Conservation of Nature stated that the single most important action needed to protect the harbour porpoise (*Phocoena phocoena*) was to reduce incidental take in gill nets and other fishing gear (Klinowska 1991).

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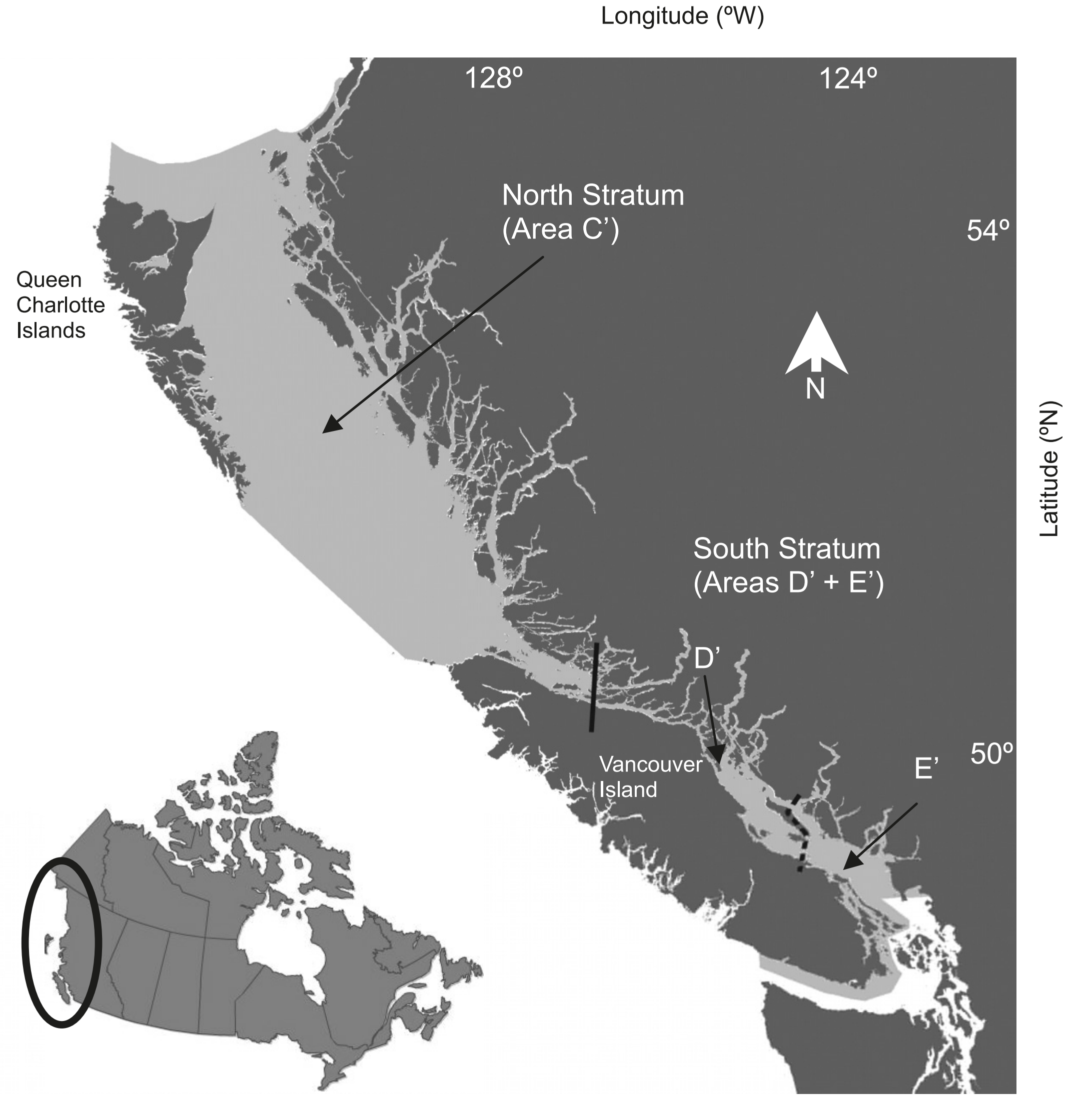
Fig. 1. Study area delineated by black oval on map of Canada (inset). The North stratum is equivalent to our area C′ (i.e., the inshore section of Fisheries and Oceans Canada's (DFO) Salmon Licensing Area C), which is composed of DFO Salmon Gillnet Management Areas 1–11 plus one-half of Area 12. The South stratum contains our areas D′ and E′ (i.e., the inshore section of DFO's Salmon Licensing Areas D and E), which are composed of the DFO Salmon Gillnet Management Areas 13–15 plus one-half of Area 12 (D′) and Areas16–20 plus Areas 28–29 (E′).



Bycatch mitigation measures are not typically driven by animal welfare needs, but rather by some balance between conservation consequences to populations and economic costs to fisheries. In eastern Canada, much effort has been directed successfully at reducing the incidental mortality of Atlantic harbour porpoise (Read and Gaskin 1988; Brodie 1995; Caswell et al. 1998). These and many other coordinated efforts evaluated estimated population sizes, stock boundaries, rates of growth, and rates of fishery-related mortality (removals) to determine whether the rate of removal was greater than the intrinsic rate of population growth (Lawson et al. 2004; Lesage et al. 2006). If so, management actions were triggered that involved the scientific, management, conservation, and fishing communities to reduce anthropogenic mortality of porpoise in Atlantic Canadian waters (Woodley 1995; Trippel et al. 1996).

On the Pacific coast of the USA and Canada, many cetacean species, including harbour porpoise, Dall's porpoise (*Phocoenoides dalli*), and Pacific white-sided dolphins (*Lagenorhynchus obliquidens*), are killed in coastal fisheries (Barlow et al. 1994; Gearin et al. 1994; Stacey et al. 1997). Mortality levels vary by location, fishery, species, season, and year; however, the harbour porpoise appears to be one of the most frequent victims of incidental mortality in commercial gear in British Columbia (BC) and Washington (Gearin et al. 1994; Baird and Guenther 1995; Stacey et al. 1997).

In Canada's Pacific region (Fig. 1), harbour porpoise are considered a species of "Special Concern" because the species is considered to be highly sensitive to human activities, is prone to becoming trapped or killed in fishing nets, and is being seen more rarely in highly developed areas, such as the waters near Victoria and Haro Strait (COSEWIC 2003). In Washington, the Inland Waters Stock (delineated as east of the northward line from Cape Flattery, Washington) is not listed as "strategic" because the species is not listed as "depleted" under the US Marine Mammal Protection Act (MMPA) or as "threatened" or "endangered" under the Endangered Species Act (NMFS 2006). However, based on the best estimates of bycatch relative to fishing effort in Washington, the National Marine Fisheries Service of the United States maintains that the total fishery mortality and serious injury for the Washington Inland Waters Stock cannot be considered to be insignificant (NMFS 2006). Understanding the population-level effects of bycatch in the inland waters is complicated by a lack of knowledge of transboundary movements of harbour porpoise and the extent to which these porpoises are subject to fishery-related mortality in BC (Hall et al. 2002; NMFS 2003; NMFS 2006).

In contrast, Dall's porpoise and Pacific white-sided dolphins are designated as "Not at Risk" by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC), because there are not thought to be any major threats to their long-term survival (Baird and Stacey 1989; Jefferson 1989). However, it should be noted that the status of these species has not been assessed since 1989 and 1990, respectively.

Dall's porpoise are known to frequent the transboundary waters of BC and Washington, whereas Pacific white-sided dolphins are only occasional visitors (Calambokidis and Baird 1994). With regard to the latter, there have been an increased number of sightings in inland BC waters since their COSEWIC listing (Morton 2000). It is also well known that fisheries technology and fishing gear have changed in that time. As with harbour porpoise, the amount of transboundary movement remains unknown.

The most recent study on small cetacean bycatch in BC (Hall et al. 2002) indicates that all three species have been caught and killed in coastal net fisheries during the time that has elapsed since the COSEWIC designations. Using direct and indirect data collection techniques, Hall et al. (2002) determined probabilities of entanglement and mortality yielding up to an estimated 100 small cetaceans killed in the commercial salmon gillnet fisheries in 2001. It was expected that different numbers of harbour porpoise, Dall's porpoise, and Pacific white-sided dolphins would be killed annually owing to changing ecological conditions and the variable nature of the fisheries (Hall et al. 2002). Previously, it has been impossible to put these estimates of bycatch in a conservation or management context, because of lack of (*i*) estimates of small cetacean abundance in BC coastal waters and (*ii*) quantitative management objectives and threshold mortality limits.

Summertime estimates of abundance have since become available for several marine mammal species in BC coastal waters. A design-unbiased survey was planned (Thomas et al. 2007) and conducted (Williams and Thomas 2007) in 2004 and 2005 and included BC inshore coastal waters (with a north–south extent from the BC–Alaska to the BC–Washington borders and an east–west extent from the mainland of BC to a line joining the north end of Vancouver Island to the south end of the Queen Charlotte Islands). Using small-boat surveys, Williams and Thomas (2007) estimated the average summertime (2004–2005) abundance of harbour porpoise in the study area to be 9120 animals (95% confidence interval, CI: 4210–19 760). Abundance estimates were also presented for Dall's porpoise (4910 animals, 95% CI: 2700–8940) and Pacific white-sided dolphins (25 900 animals, 95% CI: 12 900–52 100). These estimates are preliminary and made two key assumptions: all animals on the trackline were detected (the so-called "g(0)=1" assumption), and animal positions were recorded prior to movement in response to the boat (the assumption of no responsive movement).

Small cetacean conservation requires understanding the consequence of fisheries bycatch at the population level and having a management framework in place that motivates action when predetermined thresholds are exceeded. Though the studies of Williams and Thomas (2007) and Hall et al. (2002) were designed with independent objectives, together, the two provide data that may facilitate an attempt at calculating bycatch mortality threshold levels for small cetaceans that would trigger mitigation measures aimed at gear entanglement reduction.

The estimation of a threshold bycatch mortality limit requires two things: (*i*) management–conservation objectives and (*ii*) a procedure for calculating the maximum level of mortality that will still achieve those objectives (Johnston et al. 2000). Management objectives for conservation have recently been articulated in quantitative terms for some marine mammal stocks in Canada (e.g., Hammill and Stenson 2007), but there is currently no generic set of national objectives for all stocks. Similarly, threshold mortality limits (allowable harm) have been calculated for marine mammal stocks as part of Recovery Potential Assessments under the Canadian Species at Risk Act, but there is currently no uniform approach to calculating mortality limits.

In the USA, cetaceans are protected under the MMPA, which triggers conservation and management action when bycatch exceeds a specific mortality limit known as potential biological removal (PBR) (Wade 1998). The calculation of the PBR limit is straightforward given an estimate of population size, and the method was designed to achieve a management objective of maintaining marine mammal populations at or above their maximum net productivity level (as legislated in the MMPA). The PBR technique was designed to be conservative; for example, as uncertainty in abundance estimates increase, the PBR limits decrease (Wade 1998). While PBR is formally entrenched in US law, the PBR approach has also been used more generally to provide guidance when assessing the sustainability of bycatch of small cetaceans in many other countries (e.g., Berggren et al. 2002; Slooten et al. 2006).

Here, we report on a preliminary attempt to calculate threshold limits to anthropogenic mortality for small cetaceans in BC coastal waters. In the absence of defined Canadian management objectives and mortality limits to protect small cetacean populations from incidental mortality in fisheries, we used the PBR approach to calculate mortality limits based on the conservation objectives of two well-established, international frameworks for conservation and management: the US MMPA; and the Agreement on the Conservation of Small Cetaceans of the Baltic and North Seas (ASCOBANS). These objectives are similar in nature to management objectives outlined for other marine mammal stocks in Canada (Hammill and Stenson 2007). We then compare the estimated mortality limits with current estimates of small cetacean bycatch to determine whether the current mortality levels are consistent with the conservation objectives set by either the MMPA or ASCOBANS. We make no value judgement as to what Canada's quantitative management objectives ought to be. Instead, we apply a commonly used procedure with two sets of quantitative objectives to assess how the BC salmon gillnet fishery measures up in light of Canada's stated management objectives to maintain favourable conservation status of cetaceans in its territorial waters.

Materials and methods

Geographic areas

We restricted our analyses to the study area of Williams and Thomas (2007), that is, inshore BC waters for which small cetacean abundance estimates exist (Fig. 1). All waters on the west coasts of Vancouver Island and the Queen Charlotte Islands were excluded. These coastal waters were subdivided on the basis of Canadian Commercial Gillnet Fishing Areas – Pacific Region for salmon (www-ops2.pac.dfo-mpo.gc.ca/ops/vrndirectory/AreaDsc.cfm). Estimation of bycatch, as related to the study area of Williams and Thomas (2007), required grouping the salmon fishery statistical licensing areas. We adjusted the boundaries of the federally recognized fishing areas known as Salmon Licensing Areas C, D, and E so that they corresponded roughly to the northern and southern strata of Williams and Thomas (2007, Fig. 1). We refer subsequently to these salmon areas as C′, D′, and E′, such that C′ = Salmon Gill Net (SGN) Areas 1–11 plus one-half of Area 12; D′ = SGN Areas 13–15 plus one-half of Area 12; and E′ = SGN Areas 16–20 plus Areas 28–29. Area C′ corresponds generally to the Queen Charlotte Basin and adjacent mainland inlets and is referred to subsequently as our North stratum. Areas D′ and E′ refer to south coast waters (Johnstone Strait, Strait of Georgia, and Strait of Juan de Fuca) plus their connecting channels and adjacent inlets. These are subsequently referred to as our South stratum.

Population structure

The population structures of small cetaceans within BC waters are currently not known. Genetic and contaminant loading evidence indicates that eastern North Pacific harbour porpoise are not panmictic (i.e., there is evidence to suggest that there is stock structure within the region) and low levels of genetic mixing occur along the west coast of North America (Calambokidis and Barlow 1991; Chivers et al. 2002; NMFS 2006). Stock boundaries may exist within BC and Washington waters; however, the number, range, relative sizes, and rate of mixing requires further investigation. Given the evidence for fine-scale stock structure in harbour porpoise, it seemed appropriate to consider three scenarios for harbour porpoise: (*i*) a single BC stock (Areas C′ + D′ + E′); (*ii*) two stocks split on a north (Area C′) – south (Areas D′ + E′) basis based on coastal geography; and (*iii*) a transboundary scenario in which the southern BC and the Washington State Inland Waters stocks formed a single stock. Stock structures of Dall's porpoise and Pacific white-sided dolphins are similarly unstudied in BC; however, there is little evidence from adjacent waters to suspect as much fine-scale stock structure in these species as is found in harbour porpoise. Therefore, we consider only the scenario of a single stock for Dall's porpoise and Pacific white-sided dolphins.

Abundance estimates

Data on small cetacean density and abundance came from a design-unbiased, systematic survey (Thomas et al. 2007) for which field methods, data analyses, and preliminary results have been previously described (Williams and Thomas 2007). We used the published information on stratum area and animal density to estimate animal abundance for northern (C′) and southern (D′ + E′) waters. This process assumed that animal density was uniform within the four survey strata of the Williams and Thomas (2007) study.

To make our abundance estimates correspond spatially to the SGN areas, we had to add the estimated number of small cetaceans in mainland inlets to their counterparts in adjacent Inside Passage waters. For the 32 mainland inlets in the fjord stratum of Thomas et al. (2007), 68.1% of the area was estimated to fall in northern (C′) waters and 31.9% in southern (D′ + E′) waters. To obtain coefficients of variation (CVs) for the combined North and South strata, we first calculated CVs for the northern and southern parts of the fjord stratum (Williams and Thomas 2007). We did this assuming that the overall variance of the fjord stratum was equal to the sum of the variances of the northern and southern parts of the fjord stratum (assumed to be independent) and assuming that the CVs of the northern and southern parts were equal (Seber 1982):

$$V_{\mathrm{fjord}} = \left(N_{\mathrm{nfjord}}\mathrm{CV}_{\mathrm{nfjord}}\right)^2 + \left(N_{\mathrm{sfjord}}\mathrm{CV}_{\mathrm{sfjord}}\right)^2$$

where V_{fjord} is the total variance; assuming that $\mathrm{CV}_{\mathrm{nfjord}} = \mathrm{CV}_{\mathrm{sfjord}}$, rearranging to solve for this CV gives

$$\mathrm{CV}_{\mathrm{nfjord}} = \mathrm{CV}_{\mathrm{sfjord}} = \sqrt{\frac{V_{\mathrm{fjord}}}{N^2_{\mathrm{nfjord}} + N^2_{\mathrm{sfjord}}}}$$

These CVs for the northern and southern parts of the fjord stratum were then converted to variances and added to the variances of the other estimates corresponding to our North and South strata to get overall variances (and CVs) for these combined abundance estimates.

Two outstanding issues with existing estimates of abundance are responsive movement and diving animals missed on the trackline. Responsive movement can introduce positive or negative bias in abundance estimates, depending on whether the animal moved to approach or avoid the vessel, respectively. Responsive movement was assessed by examining the behaviour data recorded at the time of the first sighting. If animals were attracted to the ship, then the proportion of sightings scored as being oriented toward the vessel would be higher than one would predict from chance alone, and vice versa. The standard methods used for assessing responsive movement issues from behavioural data are described by Palka and Hammond (2001).

The problem of missing animals on the trackline ($g(0) < 1$) is difficult to assess from a small boat. Many methods have been developed to estimate $g(0)$ using double platform experiments on large ships, but isolating independent observer platforms on a small vessel is problematic. Until we can address this issue in the field, we conducted a literature review to identify estimates of g(0) that other researchers have calculated for harbour porpoise on shipboard surveys.

Calculation of mortality limits using default methods for PBR and tuning simulations

PBR

Under the PBR procedure, the limit to removals (i.e., anthropogenic mortality) for a management area is calculated using a relatively simple equation and a current estimate of absolute abundance (Wade 1998):

$$(1) \qquad \mathrm{PBR} = N_{\min}\frac{1}{2}R_{\max}F$$

where PBR is the potential biological removal limit, $N_{\min}$ is the minimum estimated number of animals, $R_{\max}$ is maximum population growth rate (i.e., at low density), and F is a recovery factor — a parameter that can be tuned so that the PBR procedure achieves specific management objectives. We assumed an $R_{\max}$ of 0.04 following Wade (1998) and because 4% is a plausible lower value for the maximum rate of increase of a harbour porpoise population (Barlow and Boveng 1991; Woodley and Read 1991; International Whaling Commission 2000). Errors in estimates of abundance from surveys are assumed to be lognormally distributed, so that $N_{\min}$ is calculated as

$$(2) \qquad N_{\min} = O_{\mathrm{abs}}\ \exp\left[Z\sqrt{\log\left(1+\mathrm{CV}_{\mathrm{abs}}^2\right)}\right]$$

where O_{abs} is a survey estimate of absolute abundance, $\mathrm{CV}_{\mathrm{abs}}$ is the coefficient of variation of this estimate, and Z is a standard normal deviate corresponding to a specified percentile (fixed at –0.842 for the 20th percentile following Wade 1998). Equation 2 assumes that O_{abs} is the median of the error distribution around O_{abs}.

Simulation model

A simulation model was used to tune the PBR procedure to specific management objectives. The model simulated a "known" small cetacean population over time, while simulating observation of this population and the implementation of the PBR procedure. Importantly, the procedure did not have knowledge of the known population; it only operated on the simulated observed data.

The model of the known population was a simple logistic model:

$$(3) \qquad N_{t+1} = N_t\left[1 + R_{\max}\left(1-\frac{N_t}{K}\right)\right] - C_t$$

where $R_{\max}$ is maximum population growth rate (assumed to be 0.04), K is population size at carrying capacity, and C_t is realized removals in year t. Realized removals were modelled as a random deviation from the set PBR limit for year t (PBR_t):

$$(4) \qquad C_t = N\left[\mathrm{PBR}_t, (\mathrm{PBR}_t\mathrm{CV}_{\mathrm{byc}})^2\right]$$

where $\mathrm{CV}_{\mathrm{byc}}$ is the coefficient of random variation in removals (assumed to be 0.5 based on our assumed CV in the rate of entanglement; see below) and $N(\mu,\sigma^2)$ is a random normal variable with expectation μ and variance σ^2. Random deviations from the PBR limit were assumed to be independent between years.

Survey estimates of absolute abundance (O_{abs_t}) were simulated every 10 years for input to the PBR procedure. Errors in these estimates were assumed to be independent between years and surveys and lognormally distributed so that

$$(5) \qquad O_{\mathrm{abs}_t} = \exp\left\{N\left[\log\left(\frac{N_t}{\sqrt{1+\mathrm{CV}_{\mathrm{abs}_t}^2}}\right), \log(1+\mathrm{CV}_{\mathrm{abs}_t}^2)\right]\right\}$$

PBR was calculated immediately after a survey for absolute abundance (eqs. 1 and 2), and this annual bycatch limit was used until the next survey.

Simulations were initialized by setting the initial number of animals in the population to a proportion of the number of animals at carrying capacity (D_0). In each individual simulation, D_0 was randomly drawn from a uniform distribution ranging from 0.05 to 1.

Tuning simulations

The PBR procedure was tuned (i.e., determined appropriate value for F) so that long-term population status achieved a given conservation objective with 95% probability. Two conservation objectives were considered.

The first was the objective used in the original development of the PBR procedure (Wade 1998): recover and (or) maintain the population at or above 50% of carrying capacity (the likely lower limit for maximum net productivity level). The second objective was that of ASCOBANS: recover and (or) maintain the population at or above 80% of carrying capacity. Survey CVs used in the tuning simulations corresponded to the actual survey CVs, which were also used in the PBR calculations described below. We ran 1000 simulations for each tuning.

PBR limit calculations

PBR limits were calculated using eqs. 1 and 2 and survey estimates of abundance for two different tunings (two objectives) and for the three population structure scenarios: (*i*) entire survey area; (*ii*) north–south strata; and (*iii*) transboundary scenario. PBR limits were also calculated allowing abundance estimates to be underestimates owing to animals being missed on the trackline. Abundance estimates were

Table 1. Harbour porpoise (*Phocoena phocoena*) abundance estimates and coefficients of variation (CV).

	N	CV	Variance
Williams and Thomas (2007) strata			
1 (Queen Charlotte Basin)	4 587	0.52	5 667 509
2 (Johnstone Strait and Discovery Passage)	0	0	0
3 (Strait of Georgia and Juan de Fuca Strait)	3 391	0.46	2 422 596
4 (Mainland inlets)	1 140	2.26	6 637 837
4a (Mainland inlets corresponding to stratum 1)	776	3.01	5 443 130
4b (Mainland inlets corresponding to strata 2–3)	364	3.01	1 194 707
Williams and Thomas (2007) whole area	9 118	0.420 892 741	14 727 941
New area			
North (stratum 1 plus corresponding inlets)	5 363	0.621 494 202	11 110 639
South (strata 2–3 plus corresponding inlets)	3 755	0.506 544 583	3 617 303
Transboundary area			
Southern BC plus Washington inland waters	14 437	0.633 236	334 292 278

divided by 0.5 and 0.8 for harbour and Dall's porpoise, respectively, which are reasonable estimates of g(0) for these species (Barlow et al. 1997). PBR limits were then recalculated with the adjusted abundance estimates. No bias was expected in estimates of Pacific white-sided dolphin abundance because of high detectability expected for these acrobatic and gregarious animals (Williams and Thomas 2007; Barlow et al. 1997). Responsive movement was addressed for Pacific white-sided dolphins in previous analyses (Williams and Thomas 2007).

Estimating bycatch entanglement and bycatch mortality

Probabilities of entanglement (P_{ent}) and mortality rates in BC were reported by Hall et al. (2002) for the commercial salmon gillnet fleet based on observer and license holder data. P_{ent} and mortality rates were regionally determined with two independent data sets: (*i*) observer data from southern BC and (*ii*) province-wide commercial license holder data (Hall et al. 2002).

We used the south coast probability of entanglement on a per-boat-day-fished (BDF) basis and mortality rate as reported by Hall et al. (2002) (n = four small cetaceans entangled in 3236 BDF with 5% observer coverage: P_{ent} = 0.0247 animal·BDF^{-1}, mortality rate of observed entangled animals = 50%). The fishing effort in BDF reported for each area in 2004 and 2005 was simply multiplied by this probability and mortality rate to yield estimates of bycatch mortality for our northern and southern strata.

Hall et al. (2002) cautioned that such extrapolations must be exercised with vigilance because of the uncertainties associated with small sample sizes and the assumptions that the 5% observer coverage was representative of the entire salmon fleet and that P_{ent} was spatially and temporally equal. For example, it is likely that the number of animals entangled per unit fishing effort is a function of animal density. Unfortunately, there are no estimates of density available for the years of the entanglement rate data with which to adjust entanglement rates by animal density. It is also important to note that these rate estimates were based on a sample size of four animals, of which two were confirmed as harbour porpoise, one Dall's porpoise, and one small cetacean for which species identification was not available. However, in the absence of finer-scale data, they remain the only available data from which to work to obtain point estimates of mortality rate.

To illustrate uncertainty surrounding the mortality estimates, we calculated two additional rates of entanglement using the upper and lower 95% confidence limits as also presented by Hall et al. (2002). In 2001, observer coverage was 5%, and total fishing effort was 3236 BDF. This corresponds to 161.8 BDF monitored, with four animals reported entangled. Setting this as a single observation in a Poisson distribution yields 95% confidence limits of 1.09–10.24 animals entangled·162 BDF^{-1}, with a CV of 0.5 (Hall et al. 2002; Zar 1996). Using these bounds as the theoretical minimum and maximum number of animals caught within the observed number of days (161.8 BDF) allows estimation through linear extrapolation to the likely range within which the actual levels of mortality fall. The lower limit was estimated using P_{ent} = 0.0067 (1.09 animals·161.8 BDF^{-1}), while the upper limit was estimated using P_{ent} = 0.0633 (10.24 animals·161.8 BDF^{-1}).

It is likely that our confidence interval based on the Poisson distribution underestimates the uncertainty in the number of animals entangled per unit of observed fishing effort. The quasi-Poisson or negative binomial distributions, which allow for overdispersion relative to the Poisson distribution, might be more appropriate. However, with so few observed events it was not possible to reliably estimate how overdispersed these data might be. Our confidence interval for the rate of entanglement should be considered a minimum confidence interval.

Results

Area-specific estimates of harbour porpoise abundance are provided (Table 1). A number of estimates of g(0) have been reported for harbour porpoise from shipboard sightings surveys. Barlow et al. (1997) reviewed g(0) estimates reported from shipboard surveys for harbour porpoise in US waters that ranged from 0.4 to 0.78, depending, inter alia, on number of observers and sighting conditions (with three observers and good sighting conditions producing the highest detection probability). Palka (2000) conducted shipboard sightings surveys for harbour porpoise in the Gulf of Maine

Table 2. Proposed limits to anthropogenic mortality (ML) for small cetaceans in inshore waters of BC, using two different management objectives and incorporating likely scenarios for bias in abundance estimates.

				ML		
Area	N (CV)	Objective	F	g(0) = 1	g(0) = 0.5	g(0) = 0.8
Harbour porpoise						
Entire coast	9 118 (0.42)	0.5	1	129	259	
		0.8	0.41	53	106	
North	5 363 (0.62)	0.5	1	66	132	
		0.8	0.43	28	57	
South	3 755 (0.51)	0.5	1	50	100	
		0.8	0.42	21	42	
BC (South)–WA	14 437 (0.31)	0.5	1	223	447	
		0.8	0.40	89	178	
Dall's porpoise						
Entire coast	4 913 (0.29)	0.5	1	77		96
		0.8	0.40	30		38
Pacific white-sided dolphin						
Entire coast	25 906 (0.35)	0.5	1	388		
		0.8	0.41	159		

Note: The objective labelled 0.5 corresponds to a conservation objective of maintaining populations at or above 50% of K with 95% confidence, which equates roughly to the default values of the potential biological removal (PBR) calculations under the US Marine Mammal Protection Act (MMPA). The objective labelled 0.8 corresponds to a conservation objective of maintaining populations at 80% of K with 95% confidence, which equates to the management objectives under the Agreement on the Conservation of Small Cetaceans of the Baltic and North Seas (ASCOBANS).

and Bay of Fundy that produced estimates of g(0) ranging from 0.25 to 0.74, depending on platform height (9 or 14 m above the sea surface), number of observers, and stratum. Clearly, our estimate of harbour porpoise abundance is negatively biased to some degree. Williams and Thomas (2007) note that their small-boat surveys, by necessity, were conducted in good sighting conditions and always with a team of three observers. We suspect that g(0) for the Williams and Thomas (2007) survey for harbour porpoise was high (~0.75). For illustrative purposes, we calculate PBR across a range of plausible values by calculating what PBR would be if g(0) were 1 or 0.5.

Applying the methods of Palka and Hammond (2001) to our behavioural data, we found no evidence that responsive movement (avoidance or attraction) was a problem for harbour or Dall's porpoises. (Previous analyses (Williams and Thomas 2007) addressed attractive movement by dolphins.) Failure to detect significant effects suggests that any bias due to unmodelled responsive movement was negligible. Harbour porpoise are known to avoid ships (Palka and Hammond 2001), so avoidance behaviour would serve to underestimate abundance. We have no evidence to support including this as a bias parameter, and any unmodelled bias would only increase PBR. We conclude that this is a small problem at most and one that can be ignored in a precautionary framework.

Estimated mortality limits

Mortality limits were relatively low because of the large CVs associated with the estimates of abundance and bycatch (Table 2). Our abundance estimates were relatively imprecise, as expected for a small-boat survey.

Table 3. Estimates of total mortality of small cetaceans from the 2001 fisheries observer coverage and 2004 and 2005 gillnet fishing effort in boat-days fished (BDF).

Salmon area	BDF	Estimated no. entangled (BDF × P_{ent})	Estimated mortality
2004			
C′	12 480	308.14	154.07
D′	1 610	39.75	19.88
E′	1 620	40	20.00
Total			194
2005			
C′	8 605	212.73	106.37
D′	1 225	30.28	15.14
E′	730	18.05	9.02
Total			131

Estimates of total bycatch in 2004 and 2005 fisheries

Observer data

Estimates of small cetacean bycatch in the 2004 and 2005 BC salmon gillnet fishery were based on the annual BDF within the survey area of Williams and Thomas (2007) and on P_{ent} and mortality rate estimated by Hall et al. (2002). Estimates of total small cetacean mortality were determined using the observer-derived P_{ent} = 0.0247 and mortality rate = 50% of Hall et al. (2002). Point estimates of bycatch (entanglement) and mortality are provided (Table 3).

To aid in presenting the uncertainty associated with these mortality point estimates, a Poisson distribution was used to

Table 4. Confidence intervals for estimates of total mortality of small cetaceans in 2004 and 2005.

Year	Total BDF	Lower P_{ent}	Lower 95% CL	Upper P_{ent}	Upper 95% CL
2004	15 710	0.0067	52.85	0.0632	497
2005	10 560	0.0067	35.53	0.0632	334

Note: BDF, boat-days fished; P_{ent}, probability of entanglement; CL, confidence limit.

Table 5. Estimates of small cetacean gillnet-fisheries-related mortality based on entire coast point estimates of total mortality (Table 3) and three scenarios incorporating uncertainty in species identification from Hall et al. 2002.

		Mortality	
Species	Percentage	2004	2005
Scenario 1			
Harbour porpoise	50	97	66
Dall's porpoise	25	49	33
Pacific white-sided dolphin	25	49	33
Scenario 2			
Harbour porpoise	50	97	66
Dall's porpoise	50	97	66
Pacific white-sided dolphin	0	0	0
Scenario 3			
Harbour porpoise	75	146	98
Dall's porpoise	25	49	33
Pacific white-sided dolphin	0	0	0

ascertain upper and lower confidence limits (shown in Table 4).

Hall et al. (2002) indicated that bycatch in the salmon gillnet fishery most often consisted of harbour porpoise, with Dall's porpoise and Pacific white-sided dolphins being caught less often. To get species-specific estimates of mortality, three scenarios were explored based on species proportions reported in Hall et al. (2002) and our point estimates (Table 5).

License holder data

An independent estimate of P_{ent} for harbour porpoise was obtained in 2001 using a province-wide questionnaire to license holders (Hall et al. 2002). From this, Hall et al. (2002) estimated a mortality rate of 47% and P_{ent} = 0.0152 per license holder per year. Lower estimates of P_{ent} and mortality resulted from this coast-wide evaluation (Table 6). This lower rate may be due to a real effect, that is, a lower entanglement rate in north coast waters than in south coast waters, or to an apparent effect, that is, lower rates of by-catch relating to self-reporting compared with independent observer-obtained data. However, supplementary data collected by Hall et al. (2002) that spanned more than 20 years supported the initial results, indicating that bycatch of small cetaceans in the BC salmon gillnet fishery is a rare event.

The estimates in Table 6 only include harbour porpoise. Recall that harbour porpoise were found to compose either 50% or 75% of by-caught cetaceans, depending on the identity of one specimen that was not identified to the species level (Hall et al. 2002). As a result, our estimate of total mortality of all small cetacean species would be in the range of 23 to 36 animals using this method. It is important to note that in any given year, fewer license holders actually fish than are registered (L. Keary, Fisheries and Oceans Canada, 3225 Stephenson Point Road, Nanaimo, BC V9T 1K3, Canada, personal communication). However, not all license holders fish equal amounts throughout any given salmon season. We present these results for illustrative purposes and to draw attention to the very preliminary nature of existing data on current estimates of fishery-related mortality. Neither the observer data nor the license holder data alone can give a robust estimate of current levels of small cetacean mortality in BC gillnet fisheries.

Table 6. Estimates of entanglement ($P_{ent} \times$ LH) and mortality of harbour porpoise using license holder (LH) data.

				Mortality	
Salmon area	No. of LH	Estimated entanglement	Estimated mortality	LCL	UCL
2004					
C	716	10.88	5		
D	288	4.38	2		
E	395	6.00	3		
Total		21	10	5	18
2005					
C	715	10.87	5		
D	293	4.45	2		
E	381	5.79	3		
Total		21	10	5	18

Note: LCL, lower confidence limit; UCL, upper confidence limit.

Table 7. Estimates of harbour porpoise mortality in British Columbia's south coast.

Year	Our estimate: all small cetaceans	NMFS 2006: harbour porpoise only	USA and Canadian waters combined
2004	40	15	55
2005	24	15	39

Transboundary implications

According to the 2006 Stock Status Report for Inland Waters of Washington, excluding the proportion of animals expected to be in Canadian waters, the estimated size of the harbour porpoise population in inland Washington waters is N = 10 682 (CV = 0.38) when corrected for availability and perception bias (NMFS 2006). This is an average from the 2002 and 2003 population estimates (NMFS 2006). The minimum population estimate, using the lower 20th percentile of a lognormal distribution, is 7841 animals (NMFS 2006). The PBR is then 63 harbour porpoise per year the using 4% default value for R_{max} and a recovery factor of 0.4 (NMFS 2006). The minimum estimated fishery-related mortality and serious injury in US waters is 15.2 harbour porpoise per year (NMFS 2006). Transboundary totals based on the BC observer data extrapolations and the US reported mortality estimates are presented (Table 7; NMFS 2006).

It should be noted that harbour porpoise likely only comprise some proportion of the total small cetacean bycatch for our south coast estimates. However, even if our small cetacean estimates represent only harbour porpoise, the combined estimates fall within our calculated mortality limits for both objectives from MMPA (ML = 223–447; Table 2) and ASCOBANS (ML = 89–178; Table 2) frameworks based on the g(0) = 1 and g(0) = 0.5 assumptions.

Discussion

The calculations presented in this paper represent a first attempt to assess sustainable limits to anthropogenic mortality for small cetaceans in BC coastal waters and are of use as Canadian policy makers develop quantitative triggers for conservation and management and estimates of allowable harm for marine mammals (Hammill and Stenson 2007; Johnston et al. 2000). The proposed limits are preliminary because we have relatively imprecise abundance estimates, no empirical estimate of the bias in our abundance estimates (although a range of likely values are considered from those reported in the literature), and incomplete fishery coverage to estimate bycatch rates. Given the preliminary nature of the estimated rates of bycatch, definitive conclusions about the current impacts of the commercial salmon gillnet fishery in BC on small cetacean populations would be premature. It is worth noting that some attributes of the BC selective salmon gillnet fishery as it is currently practiced might lead to lower bycatch and mortality rates than might otherwise be expected. For example, BC gillnet fishery openings tend to be spatially and temporally limited. Openings are relatively short (on the order of days, rather than weeks or months), and vessels generally remain in the vicinity of their nets, thereby allowing for a prompt response to any observed entanglement event. Generally, it is reasonable to state that the extent to which small cetaceans use the specific regions targeted by fisheries is not well studied in BC.

One parameter that has large potential to influence our results and interpretation is the live-release rate of approximately 50% that we use (Hall et al. 2002). Hall et al. (2002) noted that observers in their study monitored four entanglement events, two of which resulted in live release. The authors further noted that license holders described the release of 10 of 19 (52.6%) harbour porpoise entangled in gill nets. High live-release rates are known from dedicated and labour-intensive programs in which responders remove porpoise that have become entrapped in herring weirs (reviewed in NMFS 2006), but this is certainly the exception to the rule. Studies of entangled porpoise refer to “surface drop-outs” (Trippel et al. 1996) to describe cases when porpoise carcasses have become disentangled from gear as it is being retrieved — but clearly, the findings of Hall et al. (2002) suggest that entanglement does not necessarily always lead to mortality. This preliminary result may be a reflection of small sample sizes, or the estimate from the observers may be biased because of atypical efforts on the part of fishermen to reduce porpoise mortality while the fishery is being observed. Alternatively, live releases may be inherent to this particular fishery in that the selective gear is generally attended, and indeed, both methods to estimate live-release rate resulted in similarly high values (Hall et al. 2002). Of course, net entanglement is likely a very stressful event for small cetaceans, and disentangled animals may die shortly after release. Directed studies to follow postrelease survivorship are needed, especially if conventional fishing practices in BC salmon gillnet fisheries can yield information that may increase the live-release rate in other fisheries.

On a finer scale, it must be noted that the observer data P_{ent} determined by Hall et al. (2002) is based solely on the south coast fishery. No comparable estimate of P_{ent} exists for the north coast waters, and as such we assumed that P_{ent} was applicable coast-wide. While the mortality limits themselves are highly reliant on the quality and quantity of data we have available for input, we see value in the approach used here.

Notwithstanding these caveats, a few key results emerge from this exercise. First, our best estimates of coast-wide bycatch levels in 2004 and 2005 were in the range of the MLs we calculated, and thus, were potentially of concern for harbour and Dall’s porpoises if (*i*) Canada’s management objectives were as precautionary as those under ASCOBANS or MMPA and (*ii*) current estimates of porpoise abundance were unbiased (Williams and Thomas 2007). Bycatch mortality was more likely to be exceeding ML for these species if the fraction of bycatch for which species identity is uncertain is composed of porpoise rather than dolphins. In contrast, almost no scenario that we considered makes it likely for current salmon gillnet fishing in BC coastal waters to be posing an unacceptable risk to Pacific white-sided dolphins at the population level. Fishing activities could be having a variety of sublethal impacts on dolphins and other cetaceans that are not considered here. Previous analyses have found that even low levels of clustered removals of key individuals in highly social odontocetes can result in fragmentation of social networks, which could lead to fitness-level effects (Williams and Lusseau 2006). Temporary habitat degradation associated with commercial fishing activities in this region has been linked to changes in killer whale feeding behaviour, habitat use, energetics, and activity budgets (Williams et al. 2006).

It should also be noted that while BC gillnet fisheries do cause anthropogenic mortality, numbers of bycaught porpoise were generally low compared with other regions. In the 2002 nearshore gillnet cod fishery in Newfoundland waters, Lawson et al. (2004) estimated that bycatch of small cetaceans, nearly all of which were harbour porpoise, was likely to be in the low thousands. Lesage et al. (2006) used similar methods to Hall et al. (2002), combining questionnaire surveys and at-sea observer programs to estimate incidental catch of harbour porpoises in the gillnet fishery of the Estuary and Gulf of St. Lawrence. Total bycatch in that region in 2000 and 2001 was estimated by the authors’ preferred method to be in the low thousands. In contrast, most scenarios that we considered provided estimates of total harbour porpoise bycatch in BC coastal gillnet fisheries in the low hundreds. While bycatch may be an order of magnitude lower in BC fisheries than in east coast counterparts, so too does porpoise abundance appear to be lower in west coast waters (Lawson et al. 2004, Lesage et al. 2006, Williams and Thomas 2007). As a result, porpoise bycatch may be problematic in all three management areas. Improved observer coverage is required in all three areas to refine entanglement rates, to reduce uncertainty, and to accurately assess whether current levels of anthropogenic mortality actually fall within our best estimates of sustainable limits. The possibility that PBR levels are reached or exceeded increases with data uncertainty.

One priority area for future field research should be to estimate g(0) directly for harbour and Dall's porpoise for ongoing shipboard sightings surveys, to assess how much bias actually exists in current estimates of abundance. A second priority will be to estimate seasonal variability in animal abundance. Third, mortality limits such as these are intended to be applied to panmictic population units (i.e., populations in which all individuals mix and interbreed freely). Applying these limits to larger units risks not achieving the objective for a distinct subunit if that subunit experiences a disproportionate amount of the bycatch (Wade 1998). Consequently, there is a need for better understanding of population structure in all three species. If small or restricted dolphin or porpoise populations exist in BC, a localized conservation concern may exist.

In terms of monitoring fisheries-related mortality, an obvious priority is the need for increased funding for fishery observer programs that include greater spatial and temporal coverage and adequate training in species identification of marine mammals. Our work highlights the need for accurate, consistent, and long-term monitoring of all fisheries that have the potential to affect marine mammals. Evaluation of the harbour porpoise bycatch-related mortality in the Bay of Fundy was, in part, facilitated with on-board observers that covered between 2.3% and 100% of vessel trips throughout the 1993–1994 fishing season (Trippel et al. 1996).

We note that Canada is currently in the process of developing quantitative conservation objectives based on the precautionary approach (Hammill and Stenson 2007) and calculating quantitative mortality limits for some marine mammal stocks, but also that no generic procedures have yet been made publicly available for application to all marine mammal stocks. In our view, one of the strengths of Canada's Species at Risk Act is its requirement that the "best available knowledge" be used to define objectives in recovery strategies. In that spirit, our first attempts to estimate mortality limits using the best available data, however sparse, are an important first step toward assessing the sustainability of fisheries bycatch for small cetacean species in Canada's Pacific region. This process has helped to identify data shortcomings, which, once addressed, may allow more rigorous analyses in future. We hope that the approach outlined here may provide a step in the right direction for the conservation of other data-poor species, which include many that are of little or no commercial value.

By design, PBR is a simple strategy for achieving management objectives with regard to anthropogenic mortality. PBR provides a maximum mortality, below which one would expect to achieve management objectives with respect to population status in the long term. The only data requirement of the procedure is an estimate of current abundance, which is updated with some frequency over time. For populations about which there are very few data, PBR is often implemented by setting most parameters (eqs. 1 and 2) to default values and the recovery factor (F) is used to adjust the performance of the procedure so that it is robust to uncertainty in estimates of population size, mortality, and current population status relative to K (Wade 1998). There are very few data available on the status and dynamics of small cetacean populations in our study area. Therefore, we accepted default PBR parameter values and conducted simulations to determine values for the recovery factor that would ensure our proposed management objectives would be achieved given the uncertainty in abundance, mortality, and population status.

In addition to the PBR approach, there are numerous alternative management strategies and procedures that can be taken with regard to managing anthropogenic removals from a population. For example, the International Whaling Commission has developed an algorithm for setting catch limits as part of its Revised Management Procedure (Cooke 1999). A more data-intensive approach is to conduct a population assessment to derive estimates or probability distributions for quantities of interest to management like population size, status, and dynamics (e.g., Hoyle and Maunder 2004). Mortality limits can then be set as some function of these estimates or distributions. Furthermore, these probability distributions can be used as a basis for simulations to test the performance of a management procedure and lend themselves easily to risk assessment and decision analysis techniques. Unfortunately, the data required to conduct such an assessment for small cetacean populations in BC do not currently exist and are not likely to exist in the near future. In the meantime, PBR is a simple management procedure that can be used until a more in-depth assessment can be conducted.

In general, there are large uncertainties involved in the estimation of the dynamics and status of marine mammal populations. For example, even given adequate funding, it is difficult to detect trends in abundance of marine mammal populations (Taylor et al. 2007). Management strategies that rely only on the detection of a statistically significant population decline risk only enacting necessary management measures once it is too late. Similarly, management triggers that rely only on point estimates of population size or status without consideration of the confidence intervals associated with those estimates engender high risk in the face of uncertainty. Management procedures such as PBR can be tested and tuned through simulation to ensure that they are robust to uncertainty and inherently minimize the risk that management objectives will not be met.

Much of fishery management centres on species from which economic gain is expected. Species not considered valuable, because they have no harvest value or are caught incidentally to those with harvest value, tend to be less well studied until some survival risk is determined. As part of a conservation framework in Canada, the Species at Risk Act provides guidance for compilation and evaluation of the best available science by COSEWIC. However, at this stage in the implementation of the legislation there is no mandate for providing the resources necessary to fill in data gaps for species where survival risk is a concern. In this sense, we see strong value for the implicit reward for science under the PBR approach (Wade 1998). In the PBR approach, the main reward is that more precise survey and bycatch estimates would likely allow higher, more appropriate mortality limits, which in turn would carry lower economic costs to those fisheries responsible for reducing bycatch. A similar reward-for-science framework would fit well within the overarching mandate of Fisheries and Oceans Canada, which "...includes responsibility for the conservation and sustainable use of Canada's fisheries resources while continuing to provide safe, effective and environmentally sound marine services that are responsive to the needs of Canadians in a global economy" (Fisheries and Oceans Canada Web site: www.dfo-mpo.gc.ca/us-nous_e.htm).

Quantitative management objectives will also facilitate species-specific, cross-border comparisons and cooperation, so that managers in Canada and the USA can better fulfil their respective mandates and reduce the risk to species whose numbers are threatened by current fishing practices in both countries. Common currency in terms of quantitative management objectives will also provide a better understanding of how management actions in one country can impact populations that spend time in the territorial waters of the adjacent country. Though addressing these topics may seem unwieldy, recent international cooperation between Canada and the US on the conservation and recovery of resident killer whales (*Orcinus orca*) indicates that transboundary, international cooperation and conservation of an endangered species is possible (Resident Killer Whale Recovery Team 2007). Indeed, although the two countries have treaties for exploited species (e.g., the Pacific Salmon Treaty from 1985 — www.psc.org/publications_psctreaty.htm; the Pacific Halibut Treaty from 1923 — www.iphc.washington.edu/HALCOM/history/1923can.htm), informal cooperation between the two countries on conservation of at-risk species that are subject only to nonconsumptive use may prove to be more successful.

If a management objective is to maintain a population at a certain level of its true, natural equilibrium size, then assessments of whether mortality limits are being exceeded should incorporate all known sources of anthropogenic mortality. In this study, we restricted our discussion to bycatch in commercial gillnet fisheries. But a similar approach can be used to assess whether any source of anthropogenic mortality is potentially problematic. Additional sources of mortality could include, for example, ship strikes, entanglement in antipredator nets around fish farms, intense acoustic trauma, or oil spills.

While our results are preliminary and hindered by currently small sample sizes and low statistical power, the approach we outline offers a quantitative and precautionary framework for Canada to meet its management objectives to maintain favourable conservation status of small cetacean populations. We also hope that our work can serve as a basis for transboundary evaluation of the status of (and threats to conservation of) small cetaceans in BC and Washington, where transboundary cooperation has greatly influenced the conservation of resident killer whale populations in recent years.

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