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Potential limits to anthropogenic mortality for harbour porpoises in the Baltic region

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Abstract

We estimated potential limits to anthropogenic mortality for harbour porpoises in the Baltic region (the Skagerrak, Kattegat, Great Belt and Little Belt Seas, the Kiel and Mecklenburg Bights, and the Baltic Sea) using conservation objectives set by the Agreement on the Conservation of Small Cetaceans in the Baltic and North Seas (ASCOBANS). Mortality limits (ML) were calculated as the product of: a minimum estimate of abundance, one-half the maximum rate of increase and an uncertainty factor. Previous models show that if anthropogenic mortality is less than ML, a depleted population should recover to more than 80% of carrying capacity, meeting the conservation objectives of ASCOBANS. Minimum estimates of by-catches exceed ML for the population structure hypothesis tested, indicating that these catches will impede recovery. The same result was also evident for other hypothetical population structures. We conclude that immediate management actions are necessary to reduce the magnitude of by-catches to meet the conservation objectives of ASCOBANS.

Keywords: Mortality limit; Harbour porpoise; Baltic; By-catch; ASCOBANS

1. Introduction

Large by-catches of harbour porpoises, *Phocoena phocoena*, in commercial fisheries have led to concern over the status of this species in recent years (Berggren, 1994; Perrin et al., 1994; HELCOM, 1996; IWC, 1996; Anon., 1997; ASCOBANS, 1997; Vinther, 1999). This is particularly true in the Baltic Sea and adjacent waters, where the harbour porpoise, although found throughout the year, is the only species of cetacean regularly encountered (Berggren, 1994; Kinze, 1994; Berggren and Arrhenius, 1995a, b). For example, in Swedish waters of the Skagerrak, Kattegat and Baltic Seas, the abundance of harbour porpoises declined drastically between the 1960s and 1980s (Berggren and Arrhenius, 1995a) with no subsequent recovery (Berggren and Arrhenius, 1995b). Porpoises have also become less common during this period in other areas of the Baltic region, including Danish (Andersen, 1982; Clausen and Andersen, 1988), Polish (Skora et al., 1988), and Finnish (Määttänen, 1990) waters.

As in other parts of its range, the primary threat to harbour porpoises in the Baltic region is by-catch in commercial fisheries. Due to data limitations and a lack of appropriate criteria with which to judge current levels of incidental mortality, it has not been possible to assess current levels of anthropogenic related mortality in commercial fisheries. Fortunately, however, the Agreement on the Conservation of Small Cetaceans in the Baltic and North Seas (ASCOBANS) recently provided criteria to assess sustainability of such by-catches. According to this Agreement: "Populations should be kept at or restored to 80% of their carrying capacity" (ASCOBANS, 1997).

The harbour porpoise is a small (2 m and 50 kg) coastal cetacean found over the continental shelves of the temperate northern hemisphere (Read, 1999). The species is relatively short lived (few animals live longer than 15 years) and attains sexual maturity at age 3 or 4, and gives birth to a single offspring every year, or every second year, in early summer (Read, 1990a, b; Lockyer and Kinze, 1999). To evaluate the biological impact of

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anthropogenic mortality on harbour porpoises, it is necessary to compare the magnitude of removals with the potential rate of population increase. Unfortunately, not enough empirical information exists on the potential rate of increase for harbour porpoise populations, so demographic models have been used to estimate this parameter and its associated uncertainty (Barlow and Boveng, 1991; Woodley and Read, 1991; Caswell et al., 1998). This information has then been used informally to formulate simple criteria to assess the status of populations, by comparing the magnitude of anthropogenic mortality (as a percentage of the estimated population abundance) with estimates of potential rate of increase (e.g. IWC, 1996). A fuller and more satisfactory approach should incorporate information on uncertainty and potential biases in estimates of abundance, mortality, rate of increase (e.g. Wade, 1998) as well as stock structure. In this paper, we calculate limits to anthropogenic mortality for harbour porpoises in the Baltic region following the method developed by Wade (1998), to determine whether current reported by-catch levels are consistent with the conservation criteria set by ASCOBANS.

2. Material and methods

2.1. Geographical areas

We define the Baltic region to include the Skagerrak, Kattegat, Great Belt and Little Belt Seas, the Kiel and Mecklenburg Bights, and the Baltic Sea (Fig. 1). Furthermore, we use the definition of the Baltic Sea formulated by Fonselius (1974), including the brackish waters to the east of the Darss (between Darss, Germany and Gedser, Denmark) and Limhamn ridges (at the inner border of Öresund between the Danish island of Sjælland and Sweden), including the Gulf of Finland and the Gulf of Bothnia to the north. The Kiel and Mecklenburg Bights are situated west of the Darss ridge between Germany and Denmark. The Little and Great Belt Seas are the waters around the Danish island of Fyn.

Fig. 1. Map showing the Skagerrak, Kattegat, Great Belt and Little Belt Seas, the Kiel and Mecklenburg Bights and the Baltic Sea. The dotted line in the Baltic Sea shows the border of the aerial survey conducted in 1995.

2.2. Population structure

The detailed population structure of harbour porpoises in the Baltic region is not fully understood. However, studies of morphology, genetics and contaminant loads have shown that harbour porpoises in the Baltic Sea are distinct from animals in the Skagerrak-Kattegat Seas (Börjesson and Berggren, 1997; Wang and Berggren, 1997; Berggren et al., 1999). Other studies have found population-level differences between porpoises from the Belt Seas and the North Sea (Kinze, 1985; Andersen, 1993), between the Kiel-Mecklenburg Bights and the North Sea (Tiedeman et al., 1996; Huggenberger, 1997), and between the Skagerrak–Kattegat Seas and the west coast of Norway (Wang and Berggren, 1997). Further, porpoises in the Kiel–Mecklenburg Bights and Baltic Sea are distinct on genetic and morphological grounds (Tiedeman et al., 1996; Huggenberger, 1997). Individual porpoises move throughout the Belt, Kattegat and Skagerrak Seas (Teilmann, 1998), lending support for the existence of a single population unit in this area.

Based on the information above, it is likely that at least three populations of harbour porpoises exist in the North Sea and the Baltic region. For the purpose of the present assessment, we adopt this hypothesis and assume the existence of two separate populations of porpoises in the Baltic region: (1) the Kiel–Mecklenburg Bights, the Belt, Kattegat and Skagerrak Seas; and (2) the Baltic Sea. However, we also consider the implications of other possible scenarios of population structure.

2.3. Abundance estimates

Estimates of abundance exist for a number of geographical areas in the North Sea and adjacent waters from the Small Cetacean Abundance Survey (SCANS) conducted in July 1994 (Hammond et al., 1995). One of these areas is in the Baltic region: the Skagerrak Sea-Kattegat Sea-Great Belt, where an abundance of 36,046 animals was estimated from ship-based surveys (Hammond et al., 1995). The precision of this estimate, as measured by the coefficient of variation (CV, the square root of the variance divided by the point estimate) was 0.34. The abundance of harbour porpoises in the Kiel–Mecklenburg Bights and the Baltic Sea was estimated during line transect aerial surveys conducted in these areas between 20 June and 21 July 1995. These surveys used the same methodology (both in track line design and to generate abundance estimates), aircraft and observers as were used in the SCANS survey (Hammond et al., 1995). The surveys were also flown during the same time of the year as the SCANS survey. Surveys in the Kiel–Mecklenburg Bights resulted in an estimate of 817 animals (CV = 0.48) and surveys in the Baltic Sea (excluding a 22 km corridor along the Polish coast, i.e. Fig. 1) yielded an estimate of 599 animals (CV = 0.57; Berggren et al., unpublished data).

The abundance estimate for the Baltic Sea was based on sightings of three groups, each containing a single animal, resulting in an estimated group size of 1.0 (Berggren et al., unpublished data). This is probably an underestimate of the true mean group size, as surveys in adjacent areas have observed larger groups during the same months of the year (Heide-Jørgensen et al., 1993; Hammond et al., 1995). Therefore, to generate more realistic abundance estimates for this area, we assumed two other values of mean group size for the purpose of calculating abundance estimates for the Baltic Sea. Mean group sizes were used from the Kiel Bight–Mecklenburg Bight 1995 survey (1.2, CV = 0.11, 10 sightings; Berggren et al., unpublished data) and from the Little Belt–Kiel Bight block from the 1994 survey (1.5, CV = 0.15, seven sightings; Hammond et al., 1995).

When these mean group size estimates were used for the Baltic Sea, an adjustment was made to the CV of the estimated group size to reflect that there only were three groups sighted in the Baltic Sea. This adjustment was made by assuming that the same distribution (and thus variance) of group sizes existed in the Baltic Sea as in the other areas. The variance of the sampling distribution of a mean is equal to the variance of the population divided by the sample size (Dixon and Massey, 1983). Therefore, the variance of the mean group size is related to the variance of group sizes in the population by this equation:

$$\mathrm{var}(\bar{p}) = \mathrm{var}(p)/n \tag{1}$$

where $\bar{p}$ is the estimated mean group size, $\mathrm{var}(\bar{p})$ is the variance in the estimated mean group size, $\mathrm{var}(p)$ is the variance in group sizes in the population, and n is the number of sightings. Solving for $\mathrm{var}(p)$ and putting into terms of the CV results in:

$$\mathrm{var}(p) = \mathrm{var}(\bar{p}) \times n \tag{2}$$

and because

$$\mathrm{var}(\bar{p}) = (\mathrm{CV}(\bar{p}) \times \bar{p})^2 \tag{3}$$

the variance in group sizes in the population can be expressed in terms of the CV of the mean group size as

$$\mathrm{var}(p) = (\mathrm{CV}(\bar{p}) \times \bar{p})^2 \times n \tag{4}$$

Once $\mathrm{var}(p)$ is estimated, the adjusted CV for the new sample size is calculated by using the value of 3 for n in the same equation, re-arranged to solve for $\mathrm{CV}(\bar{p})$:

$$\frac{\mathrm{CV}(\bar{p}) = \mathrm{SQRT}(\mathrm{var}(p)/3)}{(\bar{p})}. \tag{5}$$

A new abundance estimate for the Baltic Sea was then calculated from the product of group abundance and the mean group size from adjacent areas. A CV for this estimate was calculated using the formula for the CV of a product of two independent estimates:

$$\mathrm{CV(abundance)}^2 = \mathrm{CV}(\bar{p})^2 + \mathrm{CV(group\ abundance)}^2. \tag{6}$$

The total abundance for the area was calculated by simply summing the individual abundance estimates. The CV of the total was calculated from the formula for calculating the CV of the sum of two independent estimates, which is to sum the variances of the individual estimates.

2.4. By-catches

Harbour porpoises are taken in a variety of fisheries throughout the year in the Baltic region (e.g. Skora et al., 1988; Berggren, 1994; Kinze, 1994; Kock and Benke, 1996). The magnitude of incidental catches of porpoises in the bottom-set gillnet fishery for cod, *Gadus morhua*, and pollock, *Pollachius pollachius*, in the Skagerrak and Kattegat Seas has been estimated from an observer programme (i.e. data collected by independent observers monitoring the fishery aboard vessels at sea) (Carlström and Berggren, 1996a, b, 1998). Preliminary results from this programme indicate that 113 harbour porpoises (95% CI = 53–173) are killed in this fishery annually in the Skagerrak Sea alone (Carlström and Berggren, 1998).

To date, no other observer programme in the Baltic region has yielded estimates of the magnitude of by-catches. Danish set-net fisheries in the North and Baltic Seas were monitored by an observer programme between 1992 and 1998, although the sampling rate in this programme was sufficient to estimate total by-catch for the North Sea only (Vinther, 1999). The only other data available on by-catches are minimum numbers of porpoises taken, yielded from collection programmes, reporting schemes, interview surveys or other anecdotal information. These minimum estimates are shown in Table 1.

In addition to the data listed in Table 1, harbour porpoises are also subjected to by-catches in Polish, Norwegian and Finnish waters. However, by-catch data from Polish waters were not included in the analyses because the available Baltic Sea abundance estimate exclude Polish coastal waters. Further, no data on by-catch levels were available from Norway or Finland, although it is likely that by-catches occur in the Norwegian Skagerrak Sea in both the cod bottom set gillnet fishery and in the mackerel, *Scomber scombrus*, driftnet fishery. In the Finnish Baltic Sea, sporadic by-catches have been reported by Määttänen (1990) and Mattsson (1995). Although porpoises used to range within the territorial waters of Russia, Lithuania, Latvia and Estonia, there is no information available on by-catches or that the species is still found in these areas.

2.5. Calculation of mortality limits

Mortality limits were calculated as described by Wade (1998), where the mortality limit is the product of three terms:

$$\mathrm{ML} = N_{\mathrm{min}} \times 1/2R_{\mathrm{max}} \times \mathrm{UF} \tag{7}$$

where ML is the mortality limit, N_{min} is a minimum abundance estimate (defined as the 20th percentile of a log-normal distribution), R_{max} is the maximum rate of increase, and UF is an uncertainty factor (in Wade, 1998, UF called F_{R}) less than or equal to 1.0. For calculations here, a value of 0.04 was used for R_{max}. This value is taken from Woodley and Read (1991) and was used rather than the values proposed by Barlow and Boveng (1991) and Caswell et al. (1998) because these latter analyses used mortality schedules that are not realistic for harbour porpoises. Further, both ASCOBANS and the Scientific Committee of the International Whaling Commission (IWC) have used this value when modelling effects of by-catch in this species (IWC, 1996; Anon., 2000).

If anthropogenic mortality is less than a ML calculated with UF = 0.4, a population should equilibrate at more than 80% of its carrying capacity (K; Fig. 8 in Wade, 1998). However, this is only true if unbiased estimates of mortality, abundance, and R_{max} are available,

Table 1
Minimum number of by-catches per year in the Baltic Sea (BS), Mecklenburg Bight (MB), Kiel Bight (KB), Great Belt (GB), Kattegat Sea (KS) and Skagerrak Sea (SS)[a]

Area	Sweden (I)	Denmark (II[b])	Germany (III)	Sum
BS	4	3		7
BS + MB[c]			1	1
KB			12	12
GB		6		6
KS	63	31	0	94
SS	20	18		38
Sum west of BS				151

[a] The roman numerals refer to the references from which data have been taken: (I) Berggren 1994; (II) Clausen and Andersen 1988; (III) Kock and Benke 1996. For definitions of by-catch and methods of data collection refer to Appendix.

[b] Data are only from August to February and do not cover a whole year.

[c] The number of by-catches reported from the "BS + MB" area is included in "Sum west of BS" and not in "BS".

and if stock structure is known well enough so that there is no possibility that multiple populations are treated as a single population. This is the not case with the information available from the Baltic region, where uncertainties exist particularly in the estimates of mortality and the possibility of significant sub-structure within the area. Wade (1998) showed that a 50% reduction of the UF would ensure that a population was robust to plausible levels of bias (i.e. it would still achieve its recovery goal). This "bias trial" was performed for the recovery goal of 50% of K, but the results should be similar for the ASCOBANS recovery goal of 80% used here. Therefore, a UF = 0.2 should approximately meet the recovery goal here under plausible levels of bias.

To check that the use of UF = 0.2 was valid, we performed identical bias trial simulations as were done in Wade (1998), but with two differences: (1) the recovery goal was 0.8 K, and (2) more values were used for the coefficient of variation (CV) of the abundance and mortality estimates. In addition to CV = 0.2 and 0.8 as in Wade (1998), case-specific values were also used which were the actual estimated CV of each abundance estimate. The simulations were run with a generalised logistic model with $R_{\max} = 0.04$, maximum net productivity level equal to 0.5 K, abundance surveys every 4 years, and assuming that mortality estimates were at a minimum biased downwards by a factor of 2.0. See Wade (1998) for details on the simulation methods. Five thousand trajectories of 200 years in length were run where mortality was restricted to the calculated ML. The time span of 200 years was chosen as this ensured that a dynamic equilibrium had been reached (i.e. the ending conditions were not dependent on the population level at the start of the trajectory). Starting population levels were randomly sampled from the range 0.2 K–K. These sets of 5000 trajectories were repeated for different values of UF to find the value that would result in exactly 95% of the trajectories ending at or above 0.8 K.

Although it is not an explicit criterion of the ASCOBANS agreement, Wade (1998) also investigated how quickly a population reduced to a low level will recover under different mortality limits. Therefore, we made similar calculations here to determine how long it will take a severely reduced population to recover under an ML designed to meet the ASCOBANS goal. For a UF = 0.2, Wade (1998) found there was a high probability that recovery time of a severely depleted population would not be delayed by more than 15% compared to the time to recovery of a population which experienced no anthropogenic mortality (Fig. 9 in Wade, 1998). This result was for the case where a population recovered from 0.05 to 0.5 K (where no biases in the data existed). Therefore, we repeated the calculations of Wade (1998) to examine the delay in recovery time from 0.05 to 0.8 K, rather than to 0.5 K.

3. Results

The minimum annual by-catch of harbour porpoises reported from the Baltic Sea was 7; in the combined area of the Kiel–Mecklenburg Bights, the Belt, Kattegat and Skagerrak Seas (west of BS) the annual by-catch was estimated to be 151 (Table 1). The calculated minimum abundance ($N_{\min}$) for harbour porpoises in the Baltic Sea was 383 (alternatively 450 or 557, using mean group sizes of 1.2 and 1.5, respectively; Tables 2 and 3) and the calculated mortality limit (ML) for this area was 1.5 (1.8 or 2.2 for mean group sizes of 1.2 and 1.5, respectively; Tables 2 and 3). The calculated $N_{\min}$ for the area west of BS was 28,245 and the associated ML was 113 (Table 2). The minimum reported by-catch exceeded the calculated ML in the Baltic Sea (by-catch 7 and ML e.g. 2.2) and in the area west of BS (by-catch 151 and ML 113). Other hypothetical population structures that can be created by combining the different areas in Tables 1 and 2 also result in minimum reported by-catches being higher than estimated ML values. For example, when the Baltic Sea was combined with all the other areas (west of BS) the minimum by-catch was 7 + 151 = 158 and the ML value was 2.2 + 113 = 115.

The simulation results confirmed that an uncertainty factor UF = 0.2 was the approximately correct value to use in calculating mortality limits. Simulations using coefficients of variation (CVs) in the abundance and mortality estimates ranging from 0.2 to 0.8 (the same range of precision considered in Wade, 1998) resulted in values of UF from 0.21 to 0.25. Case specific simulations were also run using the actual estimated CVs (see earlier). The only estimated mortality CV was 0.27, but information on mortality came from a variety of sources (Table 1; Appendix A), so we considered CVs for the mortality estimate ranging from 0.27 to 0.8. For the Baltic Sea, using an abundance CV of 0.62, the resulting values of UF were 0.24 for both mortality CVs. Similarly, for the west of BS area, using an abundance CV of 0.33, the resulting values of UF were 0.22 for both mortality CVs (Fig. 2). The more specific calculations made here thus ranged from 0.22 to 0.24, slightly higher than the value of 0.20 used from Wade (1998). Using the case-specific value of UF = 0.24 for the BS area resulted in a ML of 2.2–2.7, depending on estimated group size (Table 3). Using the case-specific value of UF = 0.22 for the west of BS area resulted in a ML of 124 (Table 2). The minimum reported by-catch still exceeded the case-specific ML in both areas.

Using a value for UF = 0.2, there was a high probability (0.95) that the delay in recovery time to 80% of carrying capacity (K) would be no more than 18–22%, given the range of CVs considered. In other words, there was a 0.95 probability that the delay in time to recover from 0.05 to 0.8 K would be no more than 22%. This was a slightly longer delay in recovery time than found by Wade (1998) when considering recovery to 0.5K; there a UF = 0.2 resulted in a delay of no more than a 15%. When the higher case-specific values were used for UF, the delay in recovery time was no more than 25% for UF = 0.24, using CVs of 0.62 and 0.80 for abundance and mortality, respectively (Fig. 3). Under this ML, designed to meet the ASCOBANS objective of recovering to 80% K, 95% of the 5000 trajectories had recovered to 0.8 K within 136 years, whereas a population with no anthropogenic mortality would have recovered to 0.8 K in 109 years. Similarly, the delay in recovery time was no more than 27% for UF = 0.22,

Table 2
Minimum abundances (N_{min}) and mortality limits (ML) calculated for the Baltic Sea (BS), Kiel Bight–Mecklenburg Bight (KB–MB), Little Belt (LB) and Skagerrak Sea–Kattegat Sea–Great Belt (SS–KS–GB)[a]

Survey areas	Group abundance	CV	Group size	CV	Abundance	CV	N_{min}	ML (UF = 0.2)	ML (UF = case specific)
BS (I)	599	0.57	1.00	0.00	599	0.57	383	1.5	1.8
KB–MB (I)	817	0.48	1.20	0.11	980	0.49	663	2.6	2.9
SS–KS–GB (II)			1.46	0.06	36,046	0.34	27,284	109	120
Sum west of BS					37,026	0.33	28,245	113	124

[a] The roman numbers refers to the references from which survey data have been compiled: (I) Berggren et al. (unpublished data), (II) Hammond et al. (1995). The case specific UFs used for the MLs are 0.24 for the Baltic Sea and 0.22 for all other areas. Note that N_{min} for "Sum west of BS" is calculated from the summed abundance and CV, and is not equal to the sum of the individual N_{min}s for each area.

Table 3
Calculated stock abundances, minimum stock abundances (N_{min}) and mortality limits (ML) for the Baltic Sea assuming mean group size of 1.2 and 1.5, respectively[a]

Stock	Group abundance	CV	Group size	CV	Abundance	CV	N_{min}	ML (UF = 0.2)	ML (UF = 0.24)
Baltic Sea	599	0.57	1.20	0.20	718	0.60	450	1.8	2.2
Baltic Sea	599	0.57	1.50	0.23	898	0.62	557	2.2	2.7

[a] Survey data from Berggren et al. (unpublished data).

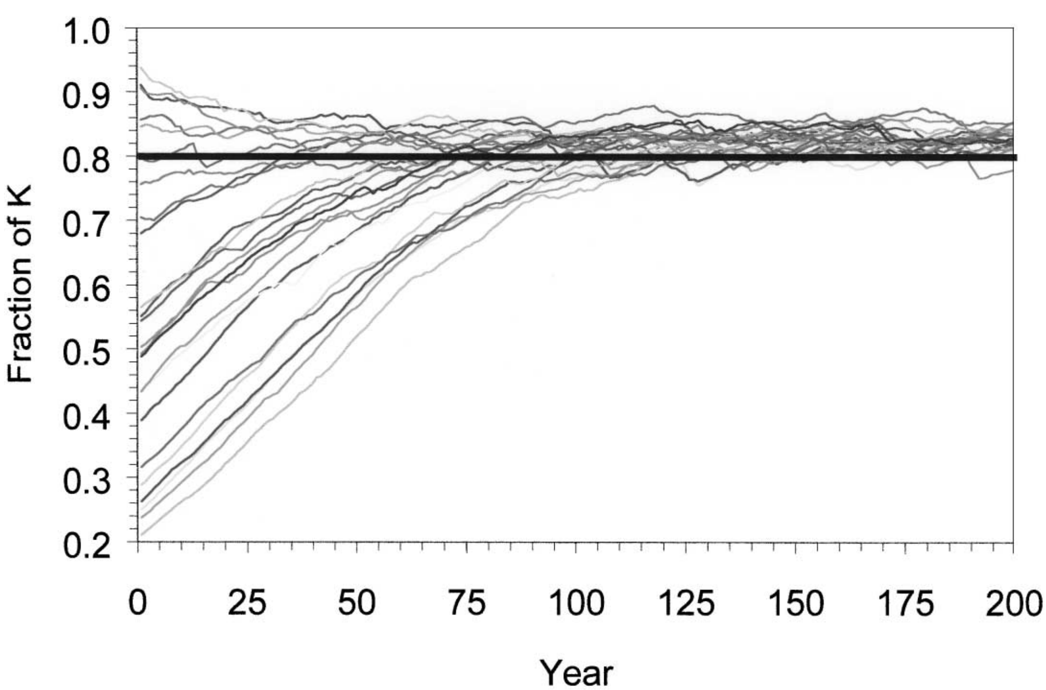

Fig. 2. Case-specific simulation results for the "west of Baltic Sea" area, using CVs of 0.33 and 0.80 for abundance and mortality, respectively, and a value for UF of 0.22. For clarity, only 25 random trajectories (out of 5000) are shown. The thick line represents the recovery goal, which is 80% of carrying capacity (K). Note that the ending population sizes after 200 years are independent of the starting population levels, which ranged from 0.2 K to K.

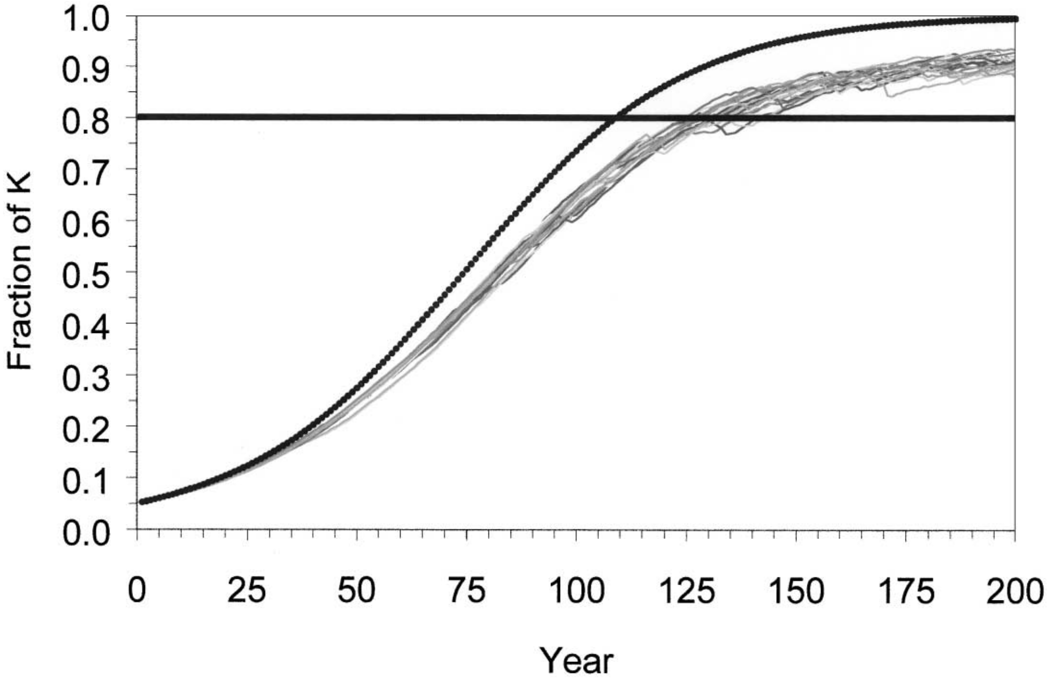

Fig. 3. Case-specific simulation results for the Baltic Sea area, using CVs of 0.62 and 0.80 for abundance and mortality, respectively, and a value for UF of 0.24. 20 random trajectories (out of 5000) are shown. The thick line represents the recovery goal, which is 80% of carrying capacity (K). The dotted line represents a model population with no anthropogenic mortality.

using CVs of 0.33 and 0.80 for abundance and mortality, respectively.

4. Discussion

Minimum estimates of by-catches (Table 1) exceeded calculated mortality limits (Tables 2 and 3) for all possible population structure scenarios, indicating that these by-catches are not consistent with the conservation objectives of ASCOBANS. This is particularly alarming because the abundance of harbour porpoise in the Baltic Sea is believed to have declined drastically between the 1960s and 1980s (Berggren and Arrhenius, 1995a, b) with no subsequent sign of recovery (Berggren and Arrhenius, 1995b; Berggren et al., unpublished data).

It is important to note that our comparisons use minimum numbers of by-catches rather than actual estimates derived from independent observer programmes where observers monitor a fishery aboard vessels at sea. These minimum numbers were yielded from collection programmes, carcass collection, reporting schemes, interview surveys or other anecdotal information which represent the absolute minimum number of specimens that were observed and/or retrieved as by-catch from fisheries in the areas covered (Table 1; Appendix A). Currently, data from observer programmes in the Baltic region only exists for the Swedish bottom set gillnet fishery for cod and pollack in the Skagerrak Sea (Carlström and Berggren, 1996a, b, 1998). To provide some perspective on the relative magnitude of estimates derived from observer programmes and carcass collection schemes, we note the estimated annual by-catch in the cod gill net fishery in the Skagerrak Sea in 1996/1997 was 113 harbour porpoises (Carlström and Berggren, 1998), but the number of animals collected in this area from all gear types was only 20 (Table 1; Berggren, 1994).

No observer programme data exists in the Baltic Sea to allow estimation of the magnitude of by-catches in this area; although of the by-caught porpoises retrieved from Swedish fisheries in the Baltic, about 50% were taken in bottom set gillnets for cod and the remaining in salmon, *Salmo salar*, driftnets (Berggren, 1994). For the cod fishery, the recorded by-catch rate from the observer programme in the Skagerrak Sea could be applied in the Baltic Sea, if corrected for lower porpoise abundance and scaled to the fishing effort in that area. The by-catch rate in the Skagerrak Sea was estimated as 32 porpoises per 10,000 km$\times$hours of fishing (Carlström and Berggren, 1996a, b). Further, the density of harbour porpoises in the Skagerrak Sea was estimated to 0.725 animals per km^2 (Hammond et al., 1995), compared to 0.0139 animals per km^2 for the Baltic Sea (Berggren et al., unpublished data). Adjusting the Skagerrak by-catch rate for the Baltic porpoise density results in 0.6 by-catches per 10,000 km$\times$hours of fishing with bottom set gillnets for cod. In the area surveyed for porpoises in the Baltic in 1995, logbook data on fishing effort measured as net length × soak time are at present available from Swedish fishermen between 1996 and 1998. During these years, the average yearly effort by Swedish fishermen using bottom set gillnets for cod in the Baltic survey area was 2,055,519 km × hours of fishing (range 1,455,242–2,396,340). This would translate to an average annual by-catch of 126 animals, which compares with the abundance estimate of 599 (alternatively 718 or 898, using mean group size of 1.2 and 1.5, respectively; Tables 2 and 3). For salmon driftnets, the other gear type in which harbour porpoise by-catches have been documented in the Swedish fisheries in the Baltic Sea, no data on by-catch rate is available. However, for a comparison, the average yearly effort with salmon driftnets set by Swedish fishermen between 1996 and 1998 in the 1995 Baltic survey area (Fig. 1) was 107,839 km × hours of fishing (range 80,750–127,570). Although there is no by-catch rate data available to extrapolate the number of porpoises possibly by-caught by this fishing effort it is perceivable that it may indicate that the situation is even more serious than described here. It is also important to note that in the surveyed area, by-catches have also been documented in fisheries by other nations (e.g. Berggren, 1994).

Bottom set gillnets for cod and driftnets for salmon account for almost all of the reported by-catches in Swedish and Polish waters of the Baltic Sea (approximately 50% in each fishery; Berggren, 1994; Anon., 1997). Given the low abundance of harbour porpoises in the Baltic Sea, a by-catch of a porpoise is a relatively rare, but significant, event for the population. Given the relative high levels of effort in the cod bottom set gillnet and the salmon driftnet fisheries in the Baltic Sea, and the small size of many of the vessels, it would be very costly and difficult, if not impossible, to acquire a reliable estimate of by-catch using independent observer programmes in the area. We believe that the funding needed for such a programme would be better spent on encouraging and subsidising new fishing practices or modifying gear to directly mitigate the by-catch problem.

The Advisory Committee to ASCOBANS concluded at its meeting in April 1999 that the only way to ascertain that the ASCOBANS conservation objectives are met in the Baltic Sea, and indeed to secure a future for the harbour porpoise population in this area, would be to phase out the current use of salmon driftnets and bottom set gillnets for cod (ASCOBANS, 1999). The Committee further stated that where gear replacement is impossible, acoustic deterrents (pingers) might also succeed in reducing by-catches in the Baltic Sea. However, it was noted that it would not be possible to directly monitor the effectiveness of pingers due to the very low by-catch rates in the area. These devices can significantly reduce the by-catch of porpoises in gill net fisheries, but the reduction is not 100% (Dawson et al., 1998; Allen et al., 1999). This is one of the main reasons why pingers have not been considered a plausible solution to by-catches of the highly endangered vaquita, *Phocoena sinus*, a porpoise endemic to the upper Gulf of California (Anon., 1999). Like the vaquita, the harbour porpoise population in the Baltic Sea is so small that it is unlikely to sustain even the low level of by-catch that would occur if pingers were used as a mitigation tool.

Although by-catches in gillnet fisheries have been identified as the most significant threat to harbour porpoises in the Baltic Sea, this population is also faced with other natural and anthropogenic threats. By-catches occur throughout the year in the Swedish waters of the Baltic Sea (Berggren, 1994) indicating that animals overwinter in the area. Porpoises that remain in the Baltic Sea during the winter risk mortality due to ice cover, which may extend over large portions of the Baltic during sever winters (Otterlind, 1976). Mass mortalities of porpoises were reported from the Baltic by Klingberg (1947) in severe winters. Further, in a recent study of organochlorines in male harbour porpoises from the Baltic Sea, PCBs were found at levels that may cause a health risk to cetaceans (individuals and on population level) according to findings in other species and geographical areas (Wagemann and Muir, 1984; Berggren et al., 1999).

Food limitation does not appear to be a factor affecting the conservation status of harbour porpoises in the Baltic region. Herring, *Clupea harengus*, and sprat, *Sprattus sprattus*, are the most important species in the diet of harbour porpoises in the Baltic Sea (Lindroth, 1962; Aarefjord et al., 1995). Both prey species have been abundant during the last two decades in the Baltic Sea (Anon., 1994) indicating that food shortages are unlikely to affect porpoises in this area.

Our results raise serious concern for the status of harbour porpoise in the Baltic Sea region. We recommend that immediate management actions be taken to reduce the magnitude of by-catches to levels that will meet the ASCOBANS conservation objectives. Furthermore, a long-term series of surveys is required to estimate absolute or relative abundance in the Baltic region, so that we may assess the long-term effectiveness of whatever mitigation measures are employed to meet these objectives. Finally, if harbour porpoise populations have been reduced to low levels relative to their pristine population size, our results indicate it will take many years before these populations recover to 80% of carrying capacity (K), the conservation objective of ASCOBANS.

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Appendix A

References from which data on by-catch have been compiled. The table shows when the data have been collected and the references' definitions of by-catch ("–" indicates that the information was not given)

Reference	Time period	Definition of by-catch	Mandatory	Reward for submitted animal[b]
Berggren (1994)	1989–1991	Collected by-caught animal	Yes[a]	150 SEK
Clausen and Andersen (1988)	Aug 1980–Feb 1981	Collected by-caught animal	–	50 DKK
Anon. (1997)	1987–1996	–	–	–
Kinze (1994)	1986–1989	Collected by-caught animal or report of by-caught animal (interview survey)	No	–
Kock and Benke (1996)	1987–1995 in Kiel Bight, 1990–1995 in Mecklenburg Bight	Collected by-caught animalor collected stranded animal with clear evidence of by-catch	No	50 DEM for by-caught animal in Kiel Bight

[a] Mandatory to report but not to submit by-caught harbour porpoise.
[b] SEK, Swedish Kronor; DKK, Danish Kroner; DEM, German Deutsche Marks.

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