Harbour porpoise (*Phocoena phocoena*) in the North Atlantic: Abundance, removals, and sustainability of removals

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ABSTRACT

The status of harbour porpoise (*Phocoena phocoena*) populations in the North Atlantic has raised numerous concerns. Although a number of factors that may be adversely affecting harbour porpoise populations have been identified, focus has been on the impact of removals, primarily due to incidental catches in fishing gear. As a result, considerable efforts have been made to determine the levels and/or impact of bycatch in a number of areas. Unfortunately, many areas remain little studied. Currently, harbour porpoise are listed as threatened or vulnerable in many parts of their range. In order to determine if the current levels of removals are sustainable, information on stock identity and seasonal movements, population parameters, abundance, and the magnitude of removals is required. Although substantial progress has been made to improve our knowledge of these parameters in the last decade, significant gaps still exist. After reviewing the available data for each sub-population in the North Atlantic, it is clear that the information required to assess the status of harbour porpoise populations is still not available for most areas. Attempts have been made to assess the status of harbour porpoise based on trends in sightings or, in areas where information on abundance and bycatch are available, on models using arbitrary criteria and/or theoretical estimates of potential population growth. Detailed case-specific population models have been proposed but are not yet available.

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INTRODUCTION

Serious concerns have been raised about the sustainability of harbour porpoise (*Phocoena phocoena*) populations in the North Atlantic. Although a number of potential limiting factors such as pollutants, habitat change or global warming have been identified (Donovan and Bjørge 1995, Aguilar and Borrell 1995, Brodie 1995, Hutchinson 1996, Teilmann and Lowry 1996, Anonymous 1999, Koschinski 2002) the primary focus has remained on the documented levels of direct mortality, primarily through incidental catches in fishing gear (Fig. 1). A number of reviews (*e.g.* Jefferson and Curry 1994, Read 1994, Donovan and Bjørge 1995, Anonymous 1998, CEC 2002) have shown that large numbers of harbour porpoise are caught in commercial fishing gear throughout their range. Based upon declining sightings and/or the perceived impacts of incidental catches, harbour porpoise populations have been classified as being in danger in many parts of their range. In Atlantic Canada, the harbour porpoise was recently reassessed as a population of Special Concern by the Committee on the Status of

Fig. 1. *Bycatch of harbour porpoises, primarily in gillnets, is a major source of mortality in some areas of the North Atlantic. (Photo: Julia Carlström)*

Endangered Wildlife in Canada (COSEWIC 2003), while the International Union for the Conservation of Nature (IUCN) considers harbour porpoise to be 'vulnerable' throughout their range (Klinowska 1991). The United States currently lists the Gulf of Maine/Bay of Fundy population as a 'strategic' species (Waring *et al.* 2001).

Considerable data are required in order to assess the status of a population and determine if the level of removals is sustainable (Donovan and Bjørge 1995, Hall and Donovan 2001). A good understanding of the stock structure and seasonal movements are required to define the area over which a population should be considered. Also, unbiased estimates of biological parameters are needed, preferably for each area and over time, to estimate population growth rates. Finally, quantitative estimates of removals and population size are necessary on the appropriate spatial scale.

In 1995, the Small Cetacean Sub-Committee of the International Whaling Commission (IWC) examined the available data required to assess the status of harbour porpoise in the North Atlantic (summarised in Donovan and Bjørge 1995). The objective of this paper is to update this review, focusing on areas where new information is available or not included in the IWC report, in order to examine the state of current knowledge required to assess the sustainability of removals on harbour porpoise populations in the North Atlantic.

STOCK IDENTITY AND DISTRIBUTION

In order to understand the impact of removals upon a population, it is imperative that the removals be applied on the correct biological scale. If 2 populations are incorrectly identified as 1, the impact of removals may be disproportionately severe on 1 population. Alternatively, if 1 population is mistakenly divided into 2, the impact of a removal may be overestimated and mitigating measures such as fishing closures may be too harsh. Identifying the correct scale can be very difficult in a species that is as mobile as harbour porpoise (for example see Read and Westgate 1997, Westgate *et al.* 1998, Teilmann 2000).

Harbour porpoise are widely distributed across the North Atlantic ranging from approximately Cape Hatteras, North Carolina in the west, north to Baffin Island and central West Greenland, Iceland, and the Faroe Islands. In the eastern Atlantic, they are found from the Barents/White Sea area south to northern West Africa (Gaskin 1984, Donovan and Bjørge 1995, Andersen 2003). A number of methods have been used to determine stock identity in the North Atlantic including life history characteristics (*e.g.* Gaskin 1984, Read and Hohn 1995), morphometrics (*e.g.* Kinze 1985, Yurick and Gaskin 1987, Gao and Gaskin 1996a, b, Borjessen and Berggren 1997), contaminant loads (*e.g.* Kleivane *et al.* 1995, Johnston 1995, Westgate 1995, Westgate *et al.* 1997, Westgate and Tolley 1999) and various genetic techniques (*e.g.* Andersen 1993, Andersen *et al.* 1995, 1997, Rosel *et al.* 1995, 1999, Wang *et al.* 1996, Wang and Berggren 1997, Walton 1997, Tolley *et al.* 1999).

In the North Atlantic, harbour porpoise can be divided into 2 separate populations, 1 in the Northwest and the other in the Northeast (Gaskin 1984, Andersen 1993, Andersen 2003). Within these populations, Gaskin (1984) identified 14 putative sub-populations, based primarily upon coincident summer distribution patterns and the assumption that harbour porpoise are confined largely to continental shelf areas. However, sighting data, satellite telemetry and records of bycatches indicate that harbour porpoise are capable of considerable movements (*e.g.* Read and Westgate 1997, Westgate *et al.* 1998, Teilmann 2000) and are not restricted to nearshore areas (Stenson and Reddin MS 1990, Rogan and Berrow 1996, Hammond *et al.* 2002). Such observations raise the possibility of exchange between putative sub-populations and highlight the need to re-evaluate their relationships using more direct methods.

After reviewing the data available up to 1995, the IWC (Donovan and Bjørge 1995) proposed 13 sub-populations. Some recent studies (*e.g.* Wang *et al.* 1996, Wang and Berggren 1997, Borjessen and Berggren 1997, Westgate *et al.*

1997, Tolley *et al.* 1999) support many of the putative sub-populations, although in many cases the differences were less apparent among males suggesting that females are more philopatric (Wang *et al.* 1996, Andersen *et al.* 1997, Tolley *et al.* 1999). However, genetic information is sparse or lacking for many areas (Andersen 2003) and there is evidence (*e.g.* Tiedemann *et al.* 1996, Gao and Gaskin 1996a, Andersen *et al.* 1997, Walton 1997,) that the population structure proposed by the IWC should be modified, particularly in the North Sea, Skagerrak, Kattegat and Baltic Sea areas. A complete review of the available information on stock structure in North Atlantic harbour porpoise is presented in Andersen (2003). For the purposes of reviewing the available information on removals, however, the 13 sub-population divisions proposed by the IWC (Donovan and Bjørge 1995) will be used in this paper.

BIOLOGICAL PARAMETERS

Information on vital parameters are available from a number of regions including the Gulf of Maine/ Bay of Fundy (Fisher and Harrison 1970, Gaskin *et al.* 1984, Read and Gaskin 1990, Read and Hohn 1995), Newfoundland (Richardson 1992, Richardson *et al.* 2003), Greenland (Kinze *et al.* MS 1990, Lockyer *et al.* 2003), Norway (Bjørge *et al.* 1991), Denmark (Møhl-Hansen 1954, Clausen and Andersen 1988, Sorensen and Kinze 1994, Lockyer and Kinze 2003), Germany (Benke *et al.* 1998), The Netherlands (Addink *et al.* 1995), Ireland (Rogan and Berrow 1995), United Kingdom (Lockyer 1995a), Spain (Lens 1997), Portugal (Sequeira 1996), and West Africa (Smeenk *et al.* 1992). The available data on population parameters of harbour porpoise in the North Atlantic are reviewed by Lockyer (2003).

Although information is available on some of the vital rates required for assessing populations (*e.g.* pregnancy rates, age of sexual maturity, etc.), nothing is known about others such as survivorship. Our ability to estimate natural mortality for most cetaceans is limited by the lack of a time series of abundance estimates, independent estimates of age structure or longitudinal studies of identified individuals. Also, the estimates of reproductive rates in most areas are based upon samples obtained as incidental catches or strandings. Many studies have found that these samples are biased towards younger animals and/or unequal sex ratios (*e.g.* IWC 1991, Richardson *et al.* 2003, Sørensen and Kinze 1994, Lockyer 1995b). Such samples can provide an indication of vital parameters that can be used as a starting point or for comparisons between samples, but caution must be used as they may exhibit unknown biases that could affect the estimates of the age structure, sex ratios and/or reproductive status of the population (Donovan and Bjørge 1995). The potential extent of some of these biases might be estimated if an area can be identified where directed catches (such as those occurring in Greenland) and bycatches occur concurrently.

ABUNDANCE

The available estimates of harbour porpoise abundance were reviewed by the IWC in 1995 (Donovan and Bjørge 1995). They presented estimates for all or part of a number of sub-populations including Gulf of Maine/Bay of Fundy (1991, 1992), Iceland (1987), North Norway/Barents Sea (1989), Kattegat and adjacent waters (1992, 1994), North Sea (1989, 1991, 1994) and Ireland and western UK (1994) (Table 1). New or updated estimates are now available for areas including the Gulf of Maine/Bay of Fundy, Gulf of St. Lawrence, North Sea and Baltic (Table 1).

Four line transect sighting surveys (1991, 1992, 1995, 1999) designed to estimate the abundance of Gulf of Maine/Bay of Fundy harbour porpoise have been carried out (Palka 1995a, 1996a, 2000). The estimates have increased, particularly between the first and second surveys, but the differences are not significant (Table 1). Inter-annual differences among the surveys may be due to a number of factors including experimental error, changes in water temperature and/or prey availability (Palka 1995b), movement among sub-populations (Waring *et al.* 2001) and improved survey design and experience. Waring *et al.* (2001) presents an average of the first 3 surveys (54,300 cv=0.14, 95% CI=41,300-71,400) as an indication of harbour

Table 1. Abundance estimates of harbour porpoises in the North Atlantic. Estimates obtained after the review by IWC in 1995 (Donovan and Bjørge 1995) are shown in bold. LT= line transect; g(0)=proportion of animals seen on the trackline; ESW=estimated strip half-width.

Abundance (cv)	95% CI	Portion of population	Month/yr	Methods	Reference
GULF OF MAINE- BAY OF FUNDY					
37,500 (0.288)	26,700-86,400	GOM/BOF	08/91	Ship LT g(0)<1	Palka 1995a
67,500 (0.231)	32,900-104,600	GOM/BOF	08/92	Ship LT g(0)<1	Palka 1995a
74,000 (0.20)	40,900-109,100	GOM/BOF	07-09/95	Ship LT g(0)<1	Palka 1996a
54,300 (0.14)	41,300-71,400	GOM/BOF	91-95	Weighted average	Waring *et al.* 2001
89,000 (0.22)	53,400-150,900	GOM/BOF	07-08/99	Ship LT g(0)<1	Palka 2000
GULF OF ST. LAWRENCE					
7,220	SE=2,340	North	08-09/95	Aerial LT g(0)=1 assumed	Kingsley and Reeves 1998
21,720	SE=8,360	North	09-10/96	Aerial LT g(0)=1 assumed	Kingsley and Reeves 1998
1,440	SE= 880	Central	08-09/95	Aerial LT g(0)=1 assumed	Kingsley and Reeves 1998
3,440	SE=1,400	South	08-09/95	Aerial LT g(0)=1 assumed	Kingsley and Reeves 1998
ICELAND					
27,000		North of 60° excluding Irminger Sea	06-07/87	NASS87-ship LT, assumed g(0)= 0.7 ESW=0.41km	Northridge MS 1995
28,514		North of 60° including Irminger Sea	06-07/87	NASS87-ship LT, assumed g(0)= 0.7 ESW=0.2044km	Sigurjónsson and Víkingsson 1997
26,843		North of 60° excluding Irminger Sea	06-07/87	NASS87-ship LT, assumed g(0)= 0.7 ESW=0.2044km	Sigurjónsson and Víkingsson 1997
N. NORWAY-BARENTS SEA					
11,000	4,790-25,200	Norwegian waters north of 66°N	07/89	NASS89-ship LT g(0)=1 assumed	Bjorge & Øien 1995
KATTEGAT AND ADJACENT WATERS					
594 (0.249)	368-967	N. Fyn	06/91	Aerial LT g(0)=1 assumed	Heide-Jorgensen *et al.* 1993
502 (0.146)	376-669	N. Fyn	06/92	Aerial LT g(0)=1 assumed	Heide-Jorgensen *et al.* 1993
207 (0.244)	132-331	Kiel Bight	06/91	Aerial LT g(0)=1assumed	Heide-Jorgensen *et al.* 1993

Abundance (cv)	95% CI	Portion of population	Month/yr	Methods	Reference
87 (0.34)	46-166	Kiel Bight	06/92	Aerial LT g(0)=1 assumed	Heide-Jorgensen *et al.* 1993
516 (0.197)	352-757	Great Belt	06/92	Aerial LT g(0)=1 assumed	Heide-Jorgensen *et al.* 1993
91 (0.384)	45-188	Little Belt	06/92	Aerial LT g(0)=1 assumed	Heide-Jorgensen *et al.* 1993
1,526 (0.13)	1,240-2,090	Great Belt	04/94	Ship LT g(0)=1 assumed	Teilmann and Lowry 1996
36,046 (0.34)	18,850-68,930	Skagerrak, Kattegat & Belt Seas (SCANS I)	07/94 LT g(0)<1	SCANS Ship	Hammond *et al.* 2002
5,262 (0.25)	3,250-8,530	Belt Seas (SCANS I')	07/94	SCANS Aerial LT g(0)<1	Hammond *et al.* 2002
588 (0.48)	240-1,440	Kiel Bight (SCANS X)	07/94	SCANS Aerial LT g(0)<1	Hammond *et al.* 2002
817	300-2,400	Kiel and Mecklenburg Bights	07/95	LT	Hiby and Lovell 1996[1]
BALTIC					
599	200-3,300	Southwest part of ICES IIId	07/95	LT	Hiby and Lovell 1996[1]
NORTH SEA					
97-486		Isle of Sylt	06/91	Aerial LT g(0)<1	Heide-Jorgensen *et al.* 1993
268,500 (0.13[2])	209,900-343,300	SCANS C -H,J,L,M,Y	07/94	SCANS Ship LT g(0)<1	Hammond *et al.* 2002
82,600	52,100-131,000	S. Norway & N. North Sea	07/89	NASS89 – Ship LT g(0)=1 assumed	Bjorge & Øien 1995
750		Dutch waters	/93	Ship LT	Smeenk
0		Channel, SCANS B	07/94	Ship LT SCANS g(0)<1	Hammond *et al.* 2002
IRELAND AND WESTERN UK					
36,280 (0.57)	12,830-102,600	Celtic shelf (SCANS A)	07/94	Ship LT SCANS g(0)<1	Hammond *et al.* 2002

[1] Cited in CEC 2002

[2] cv is slightly underestimated due to non-independence among survey blocks. See Hammond *et al.* (2002) for true cv of individual blocks.

porpoise abundance in the early 1990s. The 1999 survey is not considered directly comparable since porpoise were observed in areas that were not previously surveyed (Palka 2000) and may account for the higher estimate seen during the most recent survey.

Kingsley and Reeves (1998) conducted 2 surveys of cetaceans in the Gulf of St. Lawrence using aerial line transect methods. In late August-early September 1995 the entire Gulf was covered while only the northern area was surveyed in late July-early August 1996. Although the surveys were designed to estimate abundance of all cetaceans in the area, harbour porpoise were identified as a priority. A total of 12,100 (SE=3,200) harbour porpoise were estimated to be present in 1995 while in the following year 21,720 (SE=8,360) were estimated for the northern area alone (Table 1). These are underestimates since they were not corrected for visibility biases. Adjusting for porpoises missed by the observers may result in estimates similar to those of the Gulf of Maine/Bay of Fundy sub-population (Kingsley and Reeves 1998).

Hammond *et al.* (1995) provided estimates of harbour porpoise abundance in the North Sea, Celtic shelf, Skagerrak, Kattegat and adjacent waters based upon the SCANS (Small Cetacean Abundance in the North Sea) shipboard and aerial surveys carried out in 1994. Hammond *et al.* (2002) revised these estimates downward slightly, with an estimate of 341,366 (cv=0.14, 95% CI 260,000-449,000) porpoise in the survey area (Table 1). Using the same methodology as in the SCANS surveys, Hiby and Lovell (1996, cited in CEC 2002) estimated abundance of harbour porpoise in the Kiel and Mecklenburg Bights (ICES Area IIIc) and southwestern Baltic (ICES Sub-divisions 24 and 25, excluding a portion along the Polish coast) during July 1995. Only low numbers of porpoise were present in these areas with estimates of 817 (95% CI 300-2,400) porpoise in the Keil and Mecklenburg Bights and 599 (95% CI 200-3,300) in the southwestern Baltic.

Abundance estimates are still not available for large areas of the North Atlantic and many sub-populations do not have even minimum estimates for portions of their range. No quantifiable estimates of harbour porpoise abundance are available for Newfoundland, Greenland, Faroe Islands, Baltic, Iberia or Northwest Africa populations. In other areas (*i.e.* Iceland, North Norway/Barents Sea) the available estimates are based on surveys designed for large cetaceans and therefore may have underestimated harbour porpoise abundance. Also, these surveys are over a decade old and should be updated. Even the SCANS surveys which provided estimates for a large area in the north-eastern Atlantic are almost a decade old. A concerted effort throughout the entire range is required in order to obtain reliable estimates of current abundance of harbour porpoise in the North Atlantic.

Estimating abundance of a wide ranging species such as the harbour porpoise presents a number of difficulties that are increased by their behaviour and low visibility during surveys. Excellent survey methods have been developed for both ship and aircraft to overcome many of the methodological problems (Barlow 1988a, b, Øien 1990, Heide-Jørgensen *et al.* 1992, Borchers *et al.* 1995, Hiby and Lovell 1995, Palka 1995a, 1996b, 2000, Polacheck 1995a, b, Northridge *et al.* 1995, Laake *et al.* 1997, Hammond *et al.* 2002) and the application of satellite telemetry technology (*e.g.* Read and Westgate 1997, Westgate *et al.* 1995, 1998) may provide us with some information on movements and diving behaviour that will allow us to improve our estimates. However, surveying areas such as Newfoundland or Greenland where harbour porpoise inhabit numerous small bays and inlets can pose immense logistical problems. In such situations it may be necessary to concentrate initial survey efforts in a limited number of areas which are known or suspected to have high abundance. Information from anecdotal sightings, bycatch, prey availability and/or oceanographic features may provide some direction as long as caution is used to identify potential biases due to observer or fishing effort.

REMOVALS

Both directed and incidental removals of harbour porpoise occur in the North Atlantic.

Historically, directed removals occurred in many parts of their range (*e.g.* Bay of Fundy, Labrador, Denmark, Faroe Islands, Greenland), but in recent years they have been confined to a few areas such as Greenland and the Faroe Islands. Incidental catches continue to occur throughout the North Atlantic and a number of excellent reviews of harbour porpoise catches in the all or part of the North Atlantic are available (*e.g.* IWC 1991, 1994a; Jefferson and Curry 1994, Bjørge *et al.* 1994, Donovan and Bjørge 1995, Anonymous 1997, CEC 2002).

Although porpoise are caught in a variety of fishing gear (including trawls, longlines, weirs, seines, etc) the vast majority occur in pelagic or bottom set ("sink") gillnets (see reviews in IWC 1994a, Read 1994, Anonymous 1998). In 1990, the IWC sponsored a workshop on the mortality of cetaceans in passive fishing nets and traps (IWC 1994a) where available information on catches for a number of species, including harbour porpoise, were compiled. In 1993, the IWC passed a resolution requesting that all member countries provide annual estimates of harbour porpoise bycatch in the North Atlantic and Baltic Sea (IWC 1994b). Subsequently, ICES (Anonymous 1994) and NAMMCO (NAMMCO 1997) passed similar resolutions to monitor levels of catches by member countries. Annual estimates of incidental catches in the North Atlantic are now compiled by a number of organisations including IWC, ICES, NAMMCO and the Agreement on the Conservation of Small Cetaceans in the Baltic and North Seas (ASCOBANS). Estimates of harbour porpoise bycatch in the North Atlantic are summarised in Table 2.

Historically, information on incidental catches were obtained from anecdotal reports, voluntary reporting schemes, interviews with fishermen and/or questionnaires. Although these sources provide valuable information that can indicate areas of significant bycatch, they suffer from a number of potential biases that can affect the usefulness of the estimates. Lien *et al.* (1994) found that estimates obtained from a number of traditional methods were influenced by the methodology used and the motivation of the fishermen. The most reliable technique for obtaining quantitative estimates of total marine mammal bycatch is through the use of an independent observer scheme covering a representative sample of the fishery (IWC 1994a, 1997; Donovan and Bjørge 1995, CEC 2002). Based on this criterion, the IWC concluded in 1990 that there were no reliable estimates of incidental mortality for any fishery in the North Atlantic (IWC 1991). Since the early 1990s however, a number of countries have initiated observer programs to provide estimates of incidental catches. A number of reviews of the methodology used to assess the magnitude of bycatches (*e.g.* Donovan and Bjørge 1995, IWC 1997, Anonymous 1998, CEC 2002) are available.

Although observer programs can provide quantitative estimates of the number of harbour porpoises incidentally caught, they must be carefully designed to ensure that the estimates of catch levels and fishing effort are accurate and representative of the fishery. They also are subject to potential biases such as those associated with unobserved fishing effort and animals that may sink before being recovered (*e.g.* Vinther 1999). Hall and Donovan (2001) review some of the potential biases affecting observer programs. Initiating the large scale observer programs required for many fisheries is costly and may be difficult for a number of fisheries where incidental catches occur. This is particularly true for small boat fisheries that cannot accommodate observers, fisheries that are comprised of numerous vessels spread over large, isolated areas, or in areas where low levels of catches must be quantified both accurately and precisely. For such fisheries, alternative methods must be developed such as automated techniques, shore-based observations or logbook programs (Anonymous 1998). In a review of a number of alternative methods (logbooks, interviews, and payments) Lien *et al.* (1994) felt that maintenance of logbooks by volunteers, followed by end-of-season *in situ* interviews could provide reasonable estimates of catch levels for many fisheries. However, it is important that any method used should include checks on the reliability of the data obtained. Also, it is important that fishermen, scientists, and fisheries managers are involved in designing and implementing any reporting program in order to ensure that the estimates are accurate, reliable and accepted.

Table 2. Estimates of harbour porpoise catches in the North Atlantic. Directed catches are shown in bold. Numbers obtained using methods other than observer programs and catch statistics indicate the actual numbers and are not extrapolated to the full fishery unless indicated.

Year	Catch	95% CI / cv	Portion of population	Method	Reference
GULF OF MAINE- BAY OF FUNDY					
Gulf of Maine Sinknet Fishery					
1990	2,900	cv=0.32		Observer Program	Bravington & Bisack 1996
1991	2,000	cv=0.35		Observer Program	Bravington & Bisack 1996
1992	1,200	cv=0.21		Observer Program	Bravington & Bisack 1996
1993	1,400	cv=0.18		Observer Program	Bravington & Bisack 1996
1994	2,100	cv=0.18		Observer Program	Bisack 1997
1995	1,400	cv=0.27		Observer Program	Bisack 1997
1996	1,200	cv=0.25		Observer Program	Waring *et al.* 2001
1997	782	cv=0.22		Observer Program	Waring *et al.* 2001
1998	332	cv=0.46		Observer Program	Rossman and Merrick 1999
1999	270	cv=0.28		Observer Program	Waring *et al.* 2001
2000	507	cv=0.37		Observer Program	NMFS Unpubl. Data
Mid-Atlantic Coastal Gillnets					
1995	103	cv=0.57		Observer Program	Waring *et al.* 2001
1996	311	cv=0.31		Observer Program	Waring *et al.* 2001
1997	572	cv=0.35		Observer Program	Waring *et al.* 2001
1998	446	cv=0.36		Observer Program	Rossman and Merrick 1999
1999	53	cv=0.49		Observer Program	Rossman and Merrick 1999
2000	21	cv=0.76		Observer Program	NMFS Unpubl. Data
US Pelagic Driftnets					
1989	0.7	cv=7.00		Observer Program	Waring *et al.*. 2001
1990	1.7	cv=2.65		Observer Program	Waring *et al.*. 2001
1991	0.7	cv=1.00		Observer Program	Waring *et al.*. 2001
1992	0.4	cv=1.00		Observer Program	Waring *et al.*. 2001
1993	1.5	cv=0.34		Observer Program	Waring *et al.*. 2001
1994	0			Observer Program	Waring *et al.*. 2001
1995	0			Observer Program	Waring *et al.*. 2001
1996	0			Observer Program	Waring *et al.*. 2001
1998	0			Observer Program	Waring *et al.*. 2001
Unknown Fishery					
1999	19			Strandings	Waring *et al.*. 2001
2000	1			Strandings	NMFS Unpubl. Data
Bay of Fundy Sink Gillnets					
1993	424	SE=224		Observer Program	Trippel *et al.* 1996a
1994	101	80-122		Observer Program	Trippel *et al.* 1996a
1995	87	N/A		Observer Program	Trippel *et al.* 1996b
1996	20	‘low’		Observer Program	DFO 1998
1997	43	‘low’		Observer Program	DFO 1998
1998	10	‘low’		Observer Program	Waring *et al.* 2001
1999	<~20	‘low’		Observer Program	Waring *et al.* 2001

Tabel 2: (con´d)

Year	Catch	95% CI / cv	Portion of population	Method	Reference
Bay of Fundy Herring Weirs					
1992	11			Collections	Waring *et al.* 2001
1993	33			Collections	Waring *et al.*. 2001
1994	13			Collections	Waring *et al.*. 2001
1995	5			Collections	Waring *et al.* 2001
1996	2			Collections	Waring *et al.* 2001
1997	2			Collections	Waring *et al.* 2001
1998	2			Collections	Waring *et al.* 2001
1999	3			Collections	Waring *et al.* 2001
GULF OF ST. LAWRENCE					
1989 -90	2,000			Questionnaires	Fontaine *et al.* 1994
NEWFOUNDLAND AND LABRADOR					
1980	243		Eastern Nfld	Reported	Lien *et al.* 1988
1982 -84	41		Eastern Nfld	Logbooks	Piatt & Nettleship 1987
1980	1,368		Newfoundland	Extrapolated to fishing enterprises from Logbooks in 1 Bay	DFO 2001
1989	1,304		Newfoundland	Extrapolated to fishing enterprises from phone interviews	DFO 2001
1990	2,852 -4,416		Newfoundland	Extrapolated to fishing enterprises from logbooks	DFO 2001
1992	2,283		Newfoundland	Extrapolated to fishing enterprises from phone interviews	DFO 2001
GREENLAND					
1972	1,500		Foreign salmon driftnets	Observer program	Lear & Christensen 1975
1990 -93	**668/yr**	Range 27-1531		Catch statistics	Teilmann & Dietz 1998
1994	**1,716**			Catch statistics	NAMMCO 1996
1995	**1,135**			Catch statistics	NAMMCO 1997a
1996	**1,824**			Catch statistics	NAMMCO 2000
1997	**1,592**			Catch statistics	NAMMCO 2001
1998	**2,131**			Catch statistics	NAMMCO 2001
1999	**1,830**			Catch statistics	NAMMCO 2002
ICELAND					
1991 -95	200/yr			Collections	Víkingsson *et al.* 2003

Tabel 2: (con´d)

Year	Catch	95% CI / cv	Portion of population	Method	Reference
FAROE ISLANDS					
Late 80s	**10-20/yr**			Interviews	Larsen 1975
1996	**3**			Catch statistics	NAMMCO 1997b
N. NORWAY-BARENTS SEA					
1988-90	139[1]		Norway	Collections	Bjørge *et al.* 1991
KATTEGAT AND ADJACENT WATERS					
1980-81	55		Denmark	Collections	Clausen & Andersen 1988
1986-89	68		Denmark	Collections	Kinze 1994
1987-95	109[2]		Germany	Collections	Knock & Benke 1996
1988-91	284		Sweden	Collections	Berggren 1994
1995	53		Small area of Sweden	Observer program	Anonymous 1998
1996-97	113/yr	53-173	Swedish Skagerrak	Observer program	Berggren *et al.* 2002
BALTIC					
1980-81	3		Danish Fishery	Collections	Clausen & Andersen 1988
1984-92	3-5/yr		Sweden	Collections	Berggren 1994
1986-99	2		Finland	Unknown	CEC 2002
1987-95	<5		Germany	Collections	Knock & Benke 1996
1990-98	42		Poland	Reports	Skóra and Kuklik 2003
NORTH SEA					
1980-81	91		Denmark	Collections	Clausen & Andersen 1988
1986-89	105[3]		Denmark	Collections	Kinze 1994
	<5/yr		Netherlands	Reports	Reijnders *et al.* 1996
1990-95	66		UK	Collections	Kirwood *et al.* 1997
1990-94	23		Germany	Collections	Kotch & Benke 1996
1994-98	6,785/yr	cv=0.12	Denmark	Observer program[4]	Vinther 1999
1987	5,322 / 6,630		Denmark	Observer program[4]	Vinther and Larsen MS 2002
1988	5,938 / 6,727		Denmark	Observer program[4]	Vinther and Larsen MS 2002
1989	4,973 / 5,230		Denmark	Observer program[4]	Vinther and Larsen MS 2002
1990	5,191 / 5,257		Denmark	Observer program[4]	Vinther and Larsen MS 2002
1991	6,312 / 6,573		Denmark	Observer program[4]	Vinther and Larsen MS 2002
1992	6,543 / 7,099		Denmark	Observer program[4]	Vinther and Larsen MS 2002
1993	6,709 / 7,421		Denmark	Observer program[4]	Vinther and Larsen MS 2002
1994	7,366 / 7,566		Denmark	Observer program[4]	Vinther and Larsen MS 2002
1995	6,737 / 7,308		Denmark	Observer program[4]	Vinther and Larsen MS 2002
1996	5,991 / 6,762		Denmark	Observer program[4]	Vinther and Larsen MS 2002

Tabel 2: (con´d)

Year	Catch	95% CI / cv	Portion of population	Method	Reference
1997	5,308 / 5,731		Denmark	Observer program[4]	Vinther and Larsen MS 2002
1998	5,206 / 4,974		Denmark	Observer program[4]	Vinther and Larsen MS 2002
1999	4,227 / 3,840		Denmark	Observer program[4]	Vinther and Larsen MS 2002
2000	4,149 / 3,266		Denmark	Observer program[4,5]	Vinther and Larsen MS 2002
2001	3,887 / 2,867		Denmark	Observer program[4,5]	Vinther and Larsen MS 2002
1995	818	674-1233	UK	Observer program	CEC 2002
1996	624	500-959	UK	Observer program	CEC 2002
1997	627	513-957	UK	Observer program	CEC 2002
1998	490	383-769	UK	Observer program	CEC 2002
1999	436	351-684	UK	Observer program	CEC 2002
IRELAND AND WESTERN UK					
1993	2,200	900-3,500	Celtic Sea	Observer program	Tregenze *et al.* 1997
1995	165	82-365	West Scotland	Observer program	CEC 2002
1996	156	74-349	West Scotland	Observer program	CEC 2002
1997	209	95-475	West Scotland	Observer program	CEC 2002
1998	45	34-83	West Scotland	Observer program	CEC 2002
1999	22	14-39	West Scotland	Observer program	CEC 2002

[1] Includes animals from southern Norway/North Sea population; 96 porpoise caught in the salmon drift net fishery

[2] 104 from Kiel Bight region, may include a small number (<5) from the Baltic, approximately 20 (± 10) porpoise estimated per year in Kiel Bight and western Baltic.

[3] Includes animals from the Skagerrak.

[4] Extrapolated from bycatch rates determined from observers 1992-2001. First estimate is based on fleet effort; second is based on landings as used by Vinther (1999)

[5] Bycatch is overestimated due to use of pingers in cod wreck fishery not accounted for (Vinther and Larsen MS 2002).

Gulf of Main/Bay of Fundy

Incidental takes of harbour porpoise in the GOM/BOF region occur in both US and Canadian sink gillnets, the mid-Atlantic winter coastal gillnet fishery and the Bay of Fundy herring weir fishery. Small numbers were also reported taken in the US Atlantic pelagic drift net fishery (average of 0.4 porpoise/yr, cv=0.34) for the period 1993-96 (Waring *et al.* 2001). This fishery was closed in 1997, but reopened briefly in 1998 without any bycatch of porpoise (Waring *et al.* 2001). In 1999 the use of drift net gear in the US North Atlantic fishery was prohibited by the National Marine Fisheries Service.

Observer programs were started in the US sink gillnet fishery in 1990 and the mid-Atlantic coastal gillnet fishery in 1993 (Bisack 1997, Waring *et al.* 2001). The observer program for the Canadian sink gillnet fishery began in 1993 (DFO 1998). Although there is no observer program for the Bay of Fundy herring weir fishery, a program to release porpoises alive from weirs was initiated in 1993. Reasonably complete estimates of the number of porpoises caught (and killed) in the weirs can be obtained from this program (Waring *et al.* 2001).

Catches in the New England gillnet fishery averaged 1,163 (cv=0.11, range 332-2,100) for the period 1994-98 while an average of 358 (cv=0.20, range 103-572) porpoises were caught in the mid-Atlantic fishery from 1995-98 (Waring *et al.* 2001, Table 2). Because the bycatch was larger than the estimated Potential Biological Removals (see explanation below), 2 'Take Reduction Teams' were created in 1996 and 1997 to recommend methods of reducing the level of bycatch (DFO 2001). Since the implementation of the take reduction plans in 1999, US fishery related mortality from this sub-population has been estimated to be 342 (cv=0.25, 95%CI 211-554) and 529 (cv=0.36, 95%CI 267-1049) in 1999 and 2000, respectively (Waring *et al.* 2001, NMFS Unpublished Data). Although the 2000 point estimate was greater than that of 1999, the difference is not significant. This decline in porpoise bycatch since 1999 was due to the actions taken under the take reduction plan (time/area closures, the use of pingers, gear changes) and fisheries management plans that reduced fishing effort (DFO 2001). Reduced fishing effort appeared to have the greatest contribution to the reduction in bycatch observed in 1999 (DFO 2001).

Over 400 porpoises were estimated to have been caught in the Canadian gillnet fishery during 1993 (Table 2) although there appears to be uncertainty about this estimate due to low observer coverage (Trippel *et al.*1996a). In 1994 and 1995, catches were in the order of 90-100 porpoises. Since 1995, catches have declined significantly (1996-1998 average = 24) due to reduced fishing effort resulting from reductions in the fishing quotas (Trippel *et al.* 1996b, DFO 1998, Table 2). Preliminary analysis suggests that total mortality in 1999 is unlikely to exceed 20 (Trippel, pers. comm. cited in Waring *et al.* 2001). Until fishing effort increases, catches are likely to remain low.

The number of licenses issued for herring weirs in the Bay of Fundy has remained fairly constant since 1985. However, the number of active weirs has been decreasing (Waring *et al.* 2001). Of 263 porpoises reported caught in herring weirs between 1992 and 1994, 57 were known to have died (Table 2). From 1995-99 all but 14 of the 217 documented porpoises caught in weirs were released alive (Reid, pers. comm. cited in Waring *et al.* 2001).

There is some evidence to suggest that porpoises taken in the mid-Atlantic winter fishery may include animals from other sub-populations. Rosel *et al.* (1999) used mitochondrial data to determine if harbour porpoise present in the mid-Atlantic states during the winter were actually part of the Gulf of Maine/Bay of Fundy sub-population. They found that the animals were likely a mixture of the summer populations from the Gulf of Maine/Bay of Fundy, Gulf of St. Lawrence and Newfoundland sub-populations. However, the relative contributions of each summer population to the winter aggregation could not be determined and some haplotypes could not be assigned to a summering population. Unfortunately, most of the samples were from stranded, young animals and so the relationship of these findings to the population structure of the porpoise caught in the winter fishery is not clear.

Gulf of St. Lawrence and Newfoundland

Although bycatches are known to occur in a number of fisheries in both the Gulf of St. Lawrence and along the coast of Newfoundland, there are no quantifiable estimates based on independent observer programmes. Substantial catches are thought to have occurred since both areas have traditionally supported large gillnet (mainly cod (*Gadus morhua*)) fisheries. Based on questionnaires sent to active fishermen in 1989 and 1990 Fontaine *et al.* (1994) estimated that catches, primarily in cod groundfish gillnets, were in the order of 2,000 animals per year in the Gulf. Additional catches likely occurred along the west coast of Newfoundland which was not included in this survey.

Available information on bycatches in Newfoundland were summarized by Lien *et al.* (1988) and DFO (2001) (Table 2). Based on logbooks and interviews, Lien estimated that bycatch of harbour porpoise was likely in the low thousands during the 1980s and early 1990s (Bjørge *et al.* 1994, DFO 2001). Unfortunately, total fishing effort in Newfoundland is very difficult to determine and these estimates are based upon reported catches by a limited number of fishermen, often in restricted areas of the province. Therefore, any available estimates of bycatch in Newfoundland must be regarded with caution (DFO 2001).

Since the early 1990s, effort in the Atlantic cod fishery has been reduced significantly in both the Gulf and Newfoundland regions. This fishery, which accounted for the majority of harbour porpoises caught in these areas (Fontaine *et al.* 1994, Lien *et al.* 1994, Read 1994, DFO 2001), was closed in 1992 off the northeast coast of Newfoundland and in 1993/94 along the south coast and in the Gulf. Incidental catches of porpoise were probably significantly reduced during these moratoria (DFO 2001). The fishery in the northern Gulf was reopened at a very low level in 1997 but restricted to longlines (which do not catch a large number of porpoises) for 1997 and 1998. Gillnet fisheries have been opened in the other areas since 1997 but at reduced levels. As fish stocks in these areas recover, fishing effort will increase, likely resulting in increased levels of bycatch of harbour porpoise unless mitigation measures are taken or alternate methods of fishing used (*e.g.* crab pots, DFO 2001).

Since 1989, observers have been asked to record all incidental catches of marine mammals as part of the Department of Fisheries and Oceans Observer Program in Newfoundland (pers. comm. D. Kulka, DFO, St. John's, Nfld). Prior to 1993, this program was restricted to large offshore trawling vessels where catches of harbour porpoise appear to be low. Since 1993 there has been a low level of coverage of the inshore fishery. Vessels included were mainly 35-65 feet in length, but some coverage of smaller vessels was obtained. These data have not been examined yet, but may provide some indication of recent mortality levels in Newfoundland waters.

Greenland

Harbour porpoise are subjected to a traditional subsistence harvest in Greenland that was estimated to average over 650 animals per year for the period 1990-1993 (Tielmann and Dietz 1998; Table 2). Although catches vary seasonally, likely reflecting seasonal occurrence in coastal waters, there was a trend towards increasing catches up to the late 1960s and a decrease since 1980 (Tielmann and Dietz 1998). Since 1994, reported catches have ranged from approximately 1,100 to 2,100 porpoises (NAMMCO 1996, 1997a, 2000, 2001, 2002; Table 2). It is not clear if recent catches represent a return to the higher catch levels observed prior to 1980 or are affected by the change from the 'Hunter's List of Game' reporting system used previously to the current method (known as ‘Piniarneq’) which has been used since 1993. It is also possible that differences among years reflect trends in local abundance and/or changes in hunting effort (Teilmann and Dietz 1998). The reported catches are likely underestimates as they have not been corrected for non-reporting and lost animals.

Historically, large numbers of harbour porpoise were caught in salmon (*Salmo salar*) drift gillnets along the west coast of Greenland. Since the porpoise were fully utilised, catches in the domestic fishery were included in the reported catch (Teilmann and Dietz 1998). Using data from observers, Lear and Christensen (1975) estimated that approximately 1,500 porpoises were killed in the foreign driftnet fishery in 1972, but these data could not be extrapolated to other years due to variations in fishing effort and gear (Christensen and Lear 1977). The large-scale foreign driftnet fishery was scaled down by the mid 1970s and ceased in 1976.

Iceland

No estimates of total bycatch in Icelandic waters are available. However, an average of 200 harbour porpoises per year were collected from fishermen between 1991 and 1995 as part of a dietary study (Víkingsson *et al.* 2003). This provides a minimum estimate of the level of catches occurring.

Faroe Islands

Low numbers of harbour porpoise are taken by hunters in the Faroe Islands. Based on interviews, Larsen (1995) reported that 10-20 animals per year were taken during the late 1980s. During and after WWII, catches were slightly higher. In recent years, the only report of porpoise having been hunted was in 1996 when 3 were taken (NAMMCO 1997b). There are no estimates of the total number of harbour porpoise caught in fishing gear, but the number is though to be low (Larsen 1995).

Northern Norway/Barents Sea

Bjørge *et al.* (1991) examined 139 porpoises incidentally caught in the Norwegian coastal gillnet and salmon drift net fisheries from 1988-1990. The majority of these were caught in the drift net fishery during the first year.

Questionnaires sent to all of the licensed fishermen indicated that catch rates were relatively high in this fishery, but Bjørge *et al.* (1991) believed that the 96 porpoises reported to have been taken in 1988 was close to the true number. The salmon driftnet fishery was banned in Norwegian waters after the 1988 season.

Bjørge *et al.* (1991) also surveyed the bottom-set gillnet fisheries in 1989 and 1990. They found that although harbour porpoise were taken, the reported catch rates were significantly lower than seen in the driftnet fishery. Therefore, they concluded that the closure of the driftnet fishery in 1988 likely reduced the level of incidental catches substantially.

Kattegat and adjacent waters
Although there are no estimates of total bycatch for most fisheries in this area, there are a number of reports of incidental catches in Danish, Swedish and German waters (IWC 1997, 1998, 1999, 2000, 2001, Berggren *et al.* 2002, Table 2). Most of these records are based on the recovery of bycaught or stranded porpoises (usually with a reward) and/or voluntary reports from fishermen. These data provide some indication of the minimum levels of bycatch, but are insufficient to allow reliable estimates of total bycatch.

Small numbers of porpoise are reported taken in Swedish cod and pollock (*Pollachius pollachius*) fisheries (Table 2). The Swedish cod gillnet fishery was monitored by observers in 1995 and 1996 although only a small (~1,500 km^2) area near Gothenburg, Sweden was covered (Anonymous 1998). A similar program was carried out from March 1996 to February 1997 to monitor catches in the cod and pollock fisheries in the Swedish Skagerrak. Preliminary results from this programme indicate that 113 (95% CI 53-173) porpoises were taken annually (Carlström and Berggren 1998, cited in Berggren *et al.* 2002). Additional bycatch probably occurred in the other set-net fisheries (*e.g.* cod, plaice (*Pleuronectes platessa*), spiny dogfish (*Squalus acanthias*) and lumpsucker (*Cyclopterus lumpus*) conducted in this area. Since 1997, effort in the Swedish set net fisheries has decreased significantly (CEC 2002) which has likely reduced the associated bycatch of porpoise.

A monitoring program to estimate the level of harbour porpoise bycatch in the set-net fisheries operating in inner Danish waters has been underway since 1995. However, due to limited sampling and bycatch, estimates of annual catches are not yet available (Vinther 1999).

Baltic
There are no organised observer programs in the Baltic. Collections of animals and/or reported bycatch from fishermen provide an indication of the level of catches, but are insufficient to allow reliable estimates of total bycatch in any area. Bycatches are reported in the Danish, Swedish, German and Polish fisheries (Table 2) but the numbers of animals taken are low (*e.g.* 3, Clausen and Andersen 1988; 3-5/yr, Berggren 1994; <5 Kock and Benke 1996), which likely reflects the low abundance of porpoise in this area. Bycatch in the Swedish salmon drift net and cod gillnet fisheries may have decreased due to the decline in fishing effort over the past twenty years (CEC 2002). Skóra and Kuklik (2003) review incidental catches of harbour porpoise in Polish waters. As in Swedish waters (Berggren 1994), the majority of porpoise catches they report occurred in the salmon drift net fishery. There are no reports of catches in Russian, Latvian or Estonian waters (CEC 2002) but sightings of porpoise in these areas are rare (Anonymous 1997, Koschinski 2002). In Finland, a bycatch monitor scheme operating from 1986-1999 reported only 2 porpoise being caught (CEC 2002)

North Sea
Large numbers of harbour porpoise were taken by hunters in Danish waters from the 14th century until 1892 and again between 1916-1919 and 1941-1944. Kinze (1995) reviewed the levels of catches taken by Danish hunters during these periods.

Incidental catches in the Danish, German, Dutch, Norwegian, French and United Kingdom (UK) North Sea fisheries have been reported based on interviews or recoveries of stranded and/or by-caught animals (IWC 1996, 1997, 1998, 1999, 2000a, 2001, CEC 2002, Table 2). Based on these collections, Clausen and Andersen (1988) estimated that up to 3,000 porpoises were taken in the wreck net fishery (*i.e.* nets set around ship wrecks) in 1980 and that 'several thousand' were taken in other Danish fisheries. Similarly, Kinze (1994) roughly estimated that the annual take from a single harbour would be approximately 750 based upon the collection of 47 porpoises from the port between 1986-1989. Although only 23 porpoises were taken by German fisheries in the North Sea from 1990-1994, a large number of "stranded" porpoise (479) were recovered during the same period (Kock and Benke 1996). This and anecdotal reports of catches lead the authors to suggest that bycatches were under reported to "a large extent". They felt that catches may be in the same order of magnitude as those along the Baltic coast (20 ± 10/yr).

Reijnders *et al.* (1996) reported that the incidental catch of harbour porpoise in Dutch waters is small, usually less than 5 per year. A low level of catch is expected since only a few Dutch fishermen use gillnets. Bjørge *et al.* (1991) examined harbour porpoise caught along the coast of southern Norway in 1988 and 1989. Highest catches were obtained from the salmon driftnet fishery that was closed in 1988. The catch rates in other Norwegian gillnet fisheries were much lower. In a review of 234 stranded harbour porpoise recovered around the coasts of England and Wales between 1990 and 1995, Kirkwood *et al.* (1997) found that 66 (38%) of the 176 porpoise for which a cause of death could be determined, died as a result of entanglement in fishing gear.

An observer program designed to estimate the level of incidental catches of harbour porpoise in the North Sea Danish set-net (cod, hake (*Merluccius merluccius*), plaice, sole (Solea solea), turbot (Scophthalmus maximus)) fisheries has been underway since 1992 (Vinther 1999, Vinther and Larson MS 2002). Over 5,500 km of nets were monitored between 1992 and 1998 and resulted in an estimated average annual bycatch (based on total fleet landings of target species) of 6,785 (cv 0.12) porpoises for the North Sea fisheries in the period 1994-1998. Bycatch rates were found to vary seasonally but were not significantly different between the 1993-1995 and 1996-1998 periods (Vinther 1999). Vinther and Larson (MS 2002) have updated these estimates incorporating additional data collected since 1998. In addition to estimating bycatch based on total landings which assumes constant catch per unit effort over the entire period of the estimates (1987-2001), they also estimated bycatch based on fleet effort. Both methods suggest a decrease in recent years from a maximum of 7,366 in 1994 (based on total fleet effort) to 4,149 and 3,887 in 2000 and 2001, respectively. The decline in bycatch was greater when total landings were used as the measure of effort (Table 2). This parallels a general decrease in fishing effort in recent years (Vinther and Larsen MS 2002, CEC 2002). Catches in 2000 and 2001 were likely lower than estimated since the mandatory use of acoustic alarms (pingers) in the Danish cod wreck fishery since 2000 was not taken into account (Vinther and Larsen MS 2002)

An observer program was initiated in 1995 to estimate porpoise mortality in the UK North Sea cod, sole, skate, and turbot set net fisheries (Simon Northridge, Sea Mammal Research Unit, Gatty Marine Laboratory. University of St. Andrews. St. Andrews, Fife KY16 8LB, pers. comm., CEC 2002). Using the number of days at sea as a measure of fishing effort, harbour porpoise bycatch was estimated to have declined from 818 (95% CI 674-1233) in 1995 to 436 (95% CI 351-684) in 1999 (Table 2). This decline appears to be due to a decrease in fishing effort (CEC 2002).

Only limited data are available on bycatch of porpoise in North Sea pelagic trawl fisheries (summarised in CEC 2002). Monitoring of the Dutch pelagic trawl fishery in the North Sea and Channel from 1992-1994 indicated that although bycatch of a number of small cetaceans was recorded, no harbour porpoise were caught. No bycatch was recorded in the UK pelagic fishery. However, only 69 hauls conducted over 73 days were observed (CEC 2002).

A number of international programs (*e.g.* ASCOBANS, BY-CARE, EPIC) are being carried out to estimate the level of incidental catches in various fisheries in the North Sea and to reduce the levels of harbour porpoise bycatch. Through these programs estimates of incidental catches in this area should improve significantly.

Ireland and western UK
An observer program for the Irish and UK hake gillnet, tangle net and wreck net fisheries in the Celtic Sea operated from August 1992 to March 1994. Of the 43 harbour porpoises caught, 42 were caught in hake nets. Based on these data, Tregenza *et al.* (1997) estimated that 2,200 (95% CI 900-3,500) porpoise were caught in 1993 although this may be an underestimate since it did not include boats smaller than 15 m or trammel netters (Tregenza *et al.* 1997). It also did not include estimates of bycatch from the French fishery in the southern Celtic Sea although this is an area where porpoise densities may be lower (Anonymous 1998).

An observer program directed towards the UK gillnet and tangle net fisheries for dogfish, crayfish and skate off the west coast of Scotland was carried out from 1995 – 1999 (Northridge pers. comm., CEC 2002). The estimated number of harbour porpoise caught in these fisheries during the 1995 – 1997 period ranged from 156 (95% CI 74-349) to 209 (95% CI 95-475). In 1998 and 1999 the estimated bycatch declined to 45 (95% CI 34-83) and 22 (95% CI 14-39), respectively. This decline appeared to be due to the collapse of the crayfish tangle net fishery in this area (CEC 2002).

Iberia and Bay of Biscay
Incidental catches of harbour porpoise in fishing gear (mainly gillnets) have been reported from the French, Spanish and Portuguese coasts (Sequeria and Ferreira 1994, Sequeira 1996, Lens 1997, IWC 2000a, 2001, CEC 2002). There are no reliable estimates of the level of catches, but the porpoise appear to be one of the most common cetaceans caught in this region (Sequeria and Ferreira 1994, Lens 1997).

Northwest Africa
Incidental catches of porpoise are reported to occur in Northwest Africa (Donovan and Bjørge 1995) but the levels are unknown.

SUSTAINABILITY OF REMOVALS

Sustainability criteria
Determining the sustainability of direct or incidental removals of harbour porpoise is difficult and criteria that might be used to classify the status of populations have been the subject of great debate (*e.g.* see IWC 1996, 1997, 1998, Hammond *et al.* 1997, Taylor *et al.* 1997, Hall and Donovan 2001, Berggren *et al.* 2002, CEC 2002). One method of monitoring the sustainability of removals in a population is by following trends in abundance. Unfortunately, with the exception of the series of 4 surveys over a relatively short time period (1991 – 1999) in the Gulf of Maine/Bay of Fundy, no series of surveys exist for harbour porpoise in the North Atlantic. Also, given the high variance associated with porpoise surveys (cvs generally in the range of 15-40%, Table 1), a significant change in abundance can occur before a sufficient number of surveys could be carried out to detect the trend (Wade 1998). Finally, in a wide ranging species such as the harbour porpoise, changes in abundance in the survey area may be due to factors other than changes in absolute abundance. Changes in distribution due to differing environmental conditions, prey abundance and/or disturbance can result in spurious trends (*e.g.* see Brodie 1995, Berggren and Arrhenius 1995, Palka 1995b) and lead to incorrect assessments of the status of a population.

Populations can also be assessed through the use of models that will allow us to estimate the impact of a given level of removals. Unfortunately, there are insufficient data available to construct such population models for any specific harbour porpoise population. In part, this is due to our inability to obtain unbiased estimates of vital parameters, particularly natural mortality. In the absence of population specific models, comparing removals to estimated population size and the *potential* rate of increase of harbour porpoise has been used as a simple criteria to assess the status of a population and an indicator of a level of mortality that may not be sustainable.

The potential rates of increase (r_{max}) have been estimated by a number of authors. Barlow and Boveng (1991) used a re-scaled human mortality schedule to arrive at an estimated maximum potential rate of increase of 9.4%. A maximum rate of 4% was estimated by Woodley and Read (1991) based upon the Himalayan Thar

(*Hemitragus jemlahicus*). Using Monte-Carlo methods to evaluate the uncertainty, Caswell *et al.* (1998) estimated that potential population growth rates greater than about 14 to18% per year were unlikely and that values of about 10% seemed much more plausible. Some authors (*e.g.* IWC 2000b, Berggren *et al.* 2002) have argued that the higher estimates were obtained using mortality schedules that are not realistic for harbour porpoise. As a result, both IWC and ASCOBANS assume a value of 0.04 for r_{max} in their models (IWC 1996, 2000b). This is the same value as the assumed default for all cetaceans used in the US stock assessments (Barlow *et al.* 1995, Waring *et al.* 2001).

In 1990, the IWC Sub-Committee on Small Cetaceans reviewed the available data on estimated rates of increase and concluded that removals should be lower than half the estimated value for r_{max}. (*i.e.* 2% of population size, IWC 1991). In 1995, the sub-committee re-examined the issue and noted that the maximum growth rate could be lower than 4% per year. Given the uncertainty associated with estimates of abundance and catch levels, they adopted a figure of 1% of the estimated abundance as a "reasonable and precautionary level beyond which to be concerned about the sustainability of anthropogenic removals" (IWC 1996). This 'warning sign' is in effect accepting a level of catch equal to 25% of the r_{max} assumed for porpoise (Caswell *et al.*1998).

An alternative approach has been used in the United States where the Marine Mammal Protection Act (MMPA) defines a maximum mortality level, termed the Potential Biological Removals (PBR), beyond which management measures must be implemented (Barlow *et al.* 1995, Wade 1998). The PBR is defined as:

$$PBR = N_{min} 0.5 r_{max} F_r$$

where N_{min} is a minimum population estimate for the stock, r_{max} is the maximum theoretical or estimated rate of increase of the stock at a small size, and F_r is a recovery factor whose value lies between 0.1 and 1.0 (Barlow *et al.* 1995). In practise N_{min} is the lower 20^{th} percentile of the abundance estimate (Wade 1998) while r_{max} is assumed to be 0.04 for cetaceans. Because the status of US harbour porpoise populations relative to the 'Optimum Sustainable Population' (see below) is unknown, a default value of 0.5 is used for Fr (Barlow *et al.* 1995, Waring *et al.* 2001) which results in a critical value that is approximately 22% of the maximum growth rate (Caswell *et al.*1998). When a recovery factor of 0.5 is used the estimate of PBR is similar to the arbitrary level of 1% of population size recommended by the IWC, the only difference being the use of N_{min} rather than the point estimate (N).

There has been considerable debate about the use of a simple ratio of the point estimates of the catch and abundance to estimate a level that raises concerns about the sustainability of removals. There is general agreement that it is unsatisfactory because it does not take into account uncertainty and potential biases in the estimates (*e.g.* IWC 1996, 1997, Hammond *et al.* 1997, Taylor *et al.* 1997, Berggren *et al.* 2002). The IWC attempted to account for some of this uncertainty when they recommended a reduction in the level of 'acceptable' catch of porpoise from 2% to 1% of the estimated population size (IWC 1996, Taylor *et al.* 1997). However, this reduction is arbitrary and the robustness of these criteria to different forms of uncertainty is unknown. The IWC is currently considering approaches that will take into account uncertainty and potential biases in a more explicit manner. Five characteristics of a criterion for classifying the status of a population have been proposed: 1) the criterion to be measured is explicitly identified; 2) the criterion must use data that are available; 3) uncertainties in estimated quantities should be accounted for; 4) uncertainties in quantities for which no estimates are available should be accounted for in robustness simulation trials; and 5) the performance of any criterion should vary predictably with uncertainties in the information available with better performance from better data (IWC 1997). Taylor *et al.* (1997) proposed the use of a PBR-like approach although the performance of this method under some conditions has been questioned (*e.g.* Hammond *et al.* 1997). Another approach that incorporates uncertainties on a case-by-case basis to estimate the probability of abundance decline over a relatively short time period has also been suggested

(Bravington *et al.* 1998). An evaluation of the usefulness of this latter approach is underway (IWC 1998, 1999)

The choice of a limit beyond which catches will be considered 'unacceptable' and concern about the status of a population raised will often depend upon the management objectives one wishes to achieve. In some circumstances the goal may be zero mortality and therefore any catch is 'unsustainable'. In other circumstances the goal may be related to obtaining and/or maintaining a specific population size or trend (see Hall and Donovan 2001 for a discussion of different management objectives). The choice of limit will also be dependent upon the time frame over which the goal is to be achieved. The objective of the PBR used by the United States is to allow each stock to reach or maintain its "optimum sustainable population", defined as a population level between carrying capacity (K) and the population size at maximum net productivity. The values assumed for the calculation of PBR are chosen to ensure that there is high (95%) probability of a population being greater than 50% of K in long-term projections (Wade 1998).

ASCOBANS defined its conservation objectives as "to restore and/or maintain biological stocks of small cetaceans at the level they would reach when there is the lowest possible anthropogenic influence", *i.e.* zero mortality. Its interim objective is "to restore populations to, or maintain them at, 80% or more of the carrying capacity" (IWC 1999) and has declared that a bycatch above 2% of the estimate of abundance should be considered as an unacceptable interaction (ASCOBANS 1997). Using a basic population model for harbour porpoise, a joint IWC-ASCOBANS working group estimated that this interim objective could be met over an infinite time horizon, assuming no uncertainty in any parameter, if the maximum annual bycatch is 1.7% of the population size. In order to account for uncertainty in the estimates of population size or catch, bycatch should be less than this to ensure a high probability of meeting the ASCOBANS objective (IWC 2000b). The advice has subsequently been adopted by ASCOBANS as an interim maximum annual removal rate for harbour porpoise (ASCOBANS 2000).

Berggren *et al.* (2002) attempted to apply the PBR approach to the ASCOBANS interim objective of recovery to 80% of the carrying capacity of a population. Using data from the Baltic, Kattegat, Skagerrak and adjacent waters, N_{min} at the lower 20[th] percentile of the log-normal population distribution, and R_{max} of 0.04 in their simulations, they found that this could be achieved using an Fr of 0.22 to 0.24. A higher Fr (~0.4) would also achieve this goal if the estimates of mortality and abundance were considered to be unbiased (Wade 1998).

The goal of the US MMPA, IWC and ASCOBANS approaches used to date is to identify maximum catch levels that raise concern about the sustainability of removals in recovering populations. They are designed to be easily understood and accessible to non-scientists. However, in some areas (*e.g.* Greenland) directed takes of harbour porpoise occur and the management objectives may differ. In these situations, very different criteria for assessing the status of a population and the impact of a given level of removals may be necessary. One such approach may be to apply a method similar to the Catch Limit Algorithm that has been developed by the IWC as part of its Revised Management Procedure.

Status of populations

Before any criteria can be applied to assess the sustainability of a given level of removals on a population, accurate estimates of current abundance and removals are needed. Unfortunately, comparing Tables 1 and 2 shows that we do not have this information for most populations in the North Atlantic. For example, without any estimates of abundance and removals, there is insufficient information to assess populations in Newfoundland, Iberia and Bay of Biscay or Northwest Africa.

Abundance estimates are available for the Gulf of St. Lawrence population but the number of porpoise caught are not known. If the estimates obtained by Fontaine *et al.* (1994) accurately reflected the removals in the late 1980s, this level may not have been sustainable. However, with the extensive fisheries closures that occurred during the mid 1990s and current low level of fishing in both the Gulf and off

Newfoundland, the recent levels of bycatch are likely to have been reduced significantly in both areas (DFO 2001).

Estimates of abundance, but not removals, are also available for Iceland and Norway. In Iceland, catches are reported to be at least 200 porpoises per year, which is less than 1% of the estimated population size. The level of catches in Norwegian waters is thought to have declined since the end of the salmon driftnet fishery, but the current level is unknown. To date, there is insufficient information to assess the status of either of these populations.

Recent catches in the Faroe Islands appear to be low. Although the population size is unknown, catches appear unlikely to be a conservation issue unless the population is extremely small.

Substantial catches occur in Greenland waters but the number of porpoise in this population is unknown. Thus, there is insufficient information to determine the status of the population.

Berggren *et al.* 2002 reviewed available data on abundance and incidental catches of harbour porpoise in the Baltic, Kattegat and adjacent waters. Although reported catches in the western Baltic (7, based on data from the 1980s and early 1990s) were only 1.2% of the population estimate, they concluded that catches were above the level estimated to meet the ASCOBAN objective of recovery to 80% of the carrying capacity. Anonymous (1997) estimated that the reported catches in the western Baltic account for approximately 0.5% to 0.8% of the estimated abundance. Unfortunately, the degree to which these reported catches reflect the actual removals is not known and the data that are available on catches and abundance are dated. The apparent decline in abundance based on sightings and the levels of catch reported in the Swedish, German, Danish and Polish fisheries raises concern about the status of this population (Berggren and Arrhenius 1995, Koschinski 2002). A thorough study of stock identity, abundance and removals is required.

Small-scale observer programs have been initiated in the Swedish Skagerrak. The catch reported (113) is only 0.3% of the 36,000 animals estimated in the Skagerrak, Kattegat and adjacent waters (Stratum I) by Hammond *et al.* (2002), but this severely underestimates removals in the total area. CEC (2002) estimated that bycatch in the Swedish Skagerrak is likely to exceed 2% of the population obtained by scaling the SCANS survey densities to the smaller area. The large number of porpoise reported taken in the Kiel Bight region by the German and Danish fisheries may be having an impact on the small population estimated for this area by Hammond *et al.* (2002). Berggren *et al.* (2002) estimated that the reported removals from the Skagerrak, Great Belt and Little Belt Seas, and the Kiel and Mecklenburg Bights (151, <0.5% of the SCANS abundance estimate) was greater than the maximum removal limit estimated applying the PBR approach to the ASCOBAN goal. However, the decline in effort reported in the Swedish set net fisheries in recent years (CEC 2002) makes it difficult to estimate the current level of removals or their impact on the status of this population. In addition, questions about stock structure within the area make it difficult to apply observed catches to the appropriate sub-population. A better understanding of the population structure, levels of bycatch and seasonal movements of porpoise in the Kattegat, Baltic and North Sea areas is needed before the impact of removals can be properly assessed.

Removals in the Danish and UK North Sea set net fisheries during the mid 1990s (Vinther 1999, CEC 2002) were approximately 3% of the estimated population size of the North Sea based on the 1994 SCANS surveys (Hammond *et al.* 2002). Since then however, estimates of incidental catches in these fisheries has decreased to approximately half due to reduced fishing effort. Even though the estimates for recent years will be slightly high because the mandatory use of pingers in some sectors of the Danish fishery was not accounted for, total removals are likely an underestimate as additional catches may occur in other fisheries in the area for which we have no information. The only abundance surveys of North Sea harbour porpoise are almost a decade old. Therefore, comparing even known levels of incidental catches to abundance estimates is of dubious validity.

Although recent estimates of abundance and bycatch are not available, concern has been raised about the level of incidental catches in the Celtic Sea sub-population. The estimated bycatch in the English and Irish gillnet fisheries in the Celtic Sea was approximately 6% of the estimated abundance in this area from the 1994 SCANS survey. Although fishing effort has decreased in both of these fleets, there are no estimates of catches in a number of other the North Sea fisheries. As a result, CEC (2002) has identified this sub-population as an area of serious concern. Current abundance estimates and levels of bycatch in a number of North Sea fisheries are required to determine if the level of removals can be sustained.

The only sub-population for which we have estimates of the current levels of incidental catches and abundance is the Gulf of Maine/Bay of Fundy. Average annual catch from 1995-98 was approximately 1,300 (Table 2, Waring *et al.* 2001) which accounted for approximately 2.4% of the mean population size from 1991-95. This catch was also well above the PBR estimated prior to 1999 (Waring *et al.* 2001). However, with the implementation of the take reduction plans and reductions in fishing effort due to decreased quotas, bycatch in 1999 and 2000 was reduced to approximately 0.5% of the 1999 population estimate. Catches in these latest years are also below the estimated PBR of 747 (Waring *et al.* 2001). Although the US government has determined that listing of harbour porpoise to the Endangered Species Act is not warranted at this time, it is still considered to be a 'strategic stock' because average annual fishery-related mortality exceeded PBR for many years prior to 1999 (Waring *et al.* 2001).

CONCLUSIONS

The most important factor limiting our ability to assess the impact of removals on harbour porpoise in the North Atlantic is simply our lack of adequate information. In almost all areas, information on abundance or removals is either lacking or out of date. There is still considerable uncertainty about the population structure of porpoise in many areas of their range. In order to assess the sustainability of removals it is imperative that efforts be made to determine stock identity and monitor fishing effort, catch levels and abundance on a regular basis. This is especially critical in areas where the fisheries are undergoing significant changes. Until this is done, we will not be able to determine the status of harbour porpoise populations in any area. In addition, we must define what is meant by the term sustainability in a biologically meaningful manner and in the context of clearly stated management objectives.

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