2.5. Impact of by-catch and implications for small cetacean populations

In 1991 the IWC reviewed the impact of by-catch in 190 regional cetacean populations (Perrin 1992). It concluded that incidental catches were clearly unsustainable in 8 populations, potentially unsustainable in 34, possibly unsustainable in 5, clearly insignificant in 12, possibly insignificant in seven and of unknown consequence in an alarming 114 of the regions assessed (Perrin 1992).

The impact of anthropogenic removal of cetaceans due to by-catch may be assessed by comparing the annual by-catch estimate to the net growth rate of the population. If the percentage of animals lost to by-catch each year exceeds the percentage by which the population grows each year, then the by-catch rate is not only unsustainable but will cause a direct decline in the abundance of the population, potentially leading to extinction (Caswell *et al.* 1996, 1998). In 1995, the IWC stated that the anthropogenic removal rate of any cetacean population should not exceed half the maximum net growth rate of the population (IWC, 1995). For harbour porpoises, it was concluded that a removal rate of 1% in any population was unsustainable and thus cause for concern.

Woodley & Read (1990) modelled the growth rate of harbour porpoise populations and suggested that anything over a 5% anthropogenic removal rate would be unsustainable. This removal rate is quite high by comparison to other estimates. Carlström & Berggren (1996) modelled the growth rate of the harbour porpoise population in Swedish waters and estimated a maximum net growth rate of 4%. In the case of the harbour porpoise, the IWC now recommends that the removal rate should not exceed, or even equal, a quarter of the net population growth rate.

Under the MMPA, the critical by-catch mortality rate in porpoise populations is set at one quarter of the net growth rate (Caswell *et al.* 1996).

This low critical mortality threshold reflects increasing evidence that harbour porpoise populations are in decline (Evans & Lane 1989; Read & Gaskin 1990; Evans & Gilbert 1991; Benke *et al.* 1993; Camphuysen 1993; Evans *et al.* 1993; Kirkwood *et al.* 1997). In the EC, porpoises are listed in Annex 2 of the Habitats and Species Directive (EC Council Directive 92/43/EEC of 21 May 1992 on the Conservation of Natural Habitats of Wild Fauna and

Flora) as “species of community interest whose conservation requires the designation of special areas of conservation”.

Although the equation appears simple, many of the data required for impact assessment are unavailable. In order for the impact of removal rates to be calculated, it is important to have valid population estimates for each species susceptible to by-catch, in each region, and to have some understanding of migration patterns, the distribution of animals between regions, and stock identities (Lowry & Teilmann 1994). The most comprehensive data on population sizes in European waters comes from the SCANS project which was conducted by the Sea Mammal Research Unit (SMRU) in 1994 (Hammond *et al.* 1995). During the survey, the abundance of all cetacean species common to European waters was estimated for the North Sea, the Baltic and Swedish Seas, the Celtic Sea and the English Channel.

Generally speaking, prior to the SCANS survey, most papers estimating by-catch rates in European waters concluded that there were insufficient data for assessment of the impact of by-catch on the cetacean populations in their study regions (Kinze 1994; Lowry & Teilmann 1994; Sequeira & Ferreira 1994; Tregenza & Collet 1998). Subsequent to 1994 however, the SCANS abundance estimates have been used as a foundation on which to base impact assessments and monitor trends in the growth or decline of cetacean populations. Nonetheless, of the few papers that have estimated annual by-catch rate in an area, even fewer can assess impact of the by-catch on the cetacean population. The data are often fragmentary and refer only to certain fisheries, thus providing only a partial picture of the entire removal rate for areas in which several fisheries operate.

Aside from the overall by-catch rate, it is also important to determine whether certain components of the population suffer disproportionate mortality. Most studies involving contact with stranded or by-caught carcasses have included external examinations of by-caught animals, and have provided estimates of the sex, age and, sometimes, reproductive status of individuals caught (Goujon *et al.* 1993; Fontaine *et al.* 1994; Hall 1994; Kraus *et al.* 1995; Couperus 1997b; Tregenza *et al.* 1997a,b; Cox *et al.* 1998; Silvani *et al.* 1999). Consequently there is a good deal of information on the composition of by-catch. Finding trends in these data is problematical however, given that data collection is often opportunistic and data are therefore often not appropriate for statistical analyses. Nonetheless, the following is a brief account of the available information on cetacean population sizes in different areas, how these data are obtained and the impact of by-catch on cetacean populations in the fishing grounds as categorised above.

2.5.1. Population survey methods

Polacheck (1989) notes that "estimating the size of a marine mammal population is not an easy task, but it is so important for general assessment purposes that a large fraction of all research on marine mammals has been dedicated to just this problem". Before the impact of by-catch induced mortality on cetacean populations can be estimated, it is necessary to obtain reliable estimates of the size of the population under investigation. Prior to embarking on population surveys, it is important to consider the goals of the survey. If a population estimate is required to monitor trends in abundance, it may be sufficient to estimate the relative abundance (Anganuzzi *et al*. 1992; Anganuzzi 1993). To determine whether the anthropogenic removal rate exceeds the net growth rate of the population however, it is necessary to obtain an absolute estimate, which is generally more difficult to achieve.

Here we offer a generic (and non-technical) overview of the different survey techniques available. A detailed guide to methodology is provided by Hiby & Hammond (1989) and Garner *et al*. (1999). The most commonly used methods include strip transects, line transects, acoustic surveys, and land-based surveys (see Alling & Whitehead 1987; Chappell *et al.* 1996). Which method is used depends largely on the geography of the area, distribution of the cetacean population, funding, time constraints and expertise available for analysis of the results.

Strip transect surveys involve travel along a predetermined course, counting all cetaceans sighted within a given distance from the trackline or sampled 'strip' (Hiby & Hammond 1989; Polacheck 1989). The speed of the sighting platform must be consistent throughout the survey (Hiby & Hammond 1989).

Line transect surveys are similar but aim to account for the fact that some animals will be submerged and thus remain uncounted, and there is no set distance within which animals can be counted. However, animals sighted during any deviation from the trackline (e.g. to obtain further information on the animals sighted, to tag animals or to obtain ID photographs) are considered as "secondary sightings" and not included in estimates. The relationship between the probability of detecting a cetacean and its distance from the observer may be referred to as the “detection function” or g(0) (Hiby & Hammond 1989; Polacheck 1989).

Acoustic surveys can be used either in isolation or to augment sighting surveys. The main advantages of acoustic surveys are that they can be conducted irrespective of light levels and weather conditions, that they can survey a larger range than visual surveys and that they can be automated, thus involving fewer personnel and lower training costs. The need for fewer personnel confers the greater advantage of decreased inter-observer variability.

Land based surveys are more like a census than a survey in that observers are stationary and wait for cetaceans to pass. This can provide an absolute abundance estimate only if all the animals in a population pass through the sighting area within the active survey period, in ideal weather conditions and in daylight hours. Such circumstances are uncommon and most cetaceans in a population will have zero probability of being detected. However, in areas with good vantage points where cetaceans come close to shore, this technique can be very efficient and cost-effective (Hiby & Hammond 1989).

Whichever methodology is applied, it is rarely possible to cover the entire range of the cetacean population under investigation[12]. Consequently it is important that the proportion of the area surveyed is representative of the population range. This usually involves spatio-temporal stratification. This effectively reduces the survey area to manageable sections, each of which can be surveyed independently within realistic time spans (e.g. 10 hours) and can be considered as a single unit (Hiby & Hammond 1989).

Each species and region surveyed may require a different survey technique and survey design may be adapted to cope with limitations on observations due to the behaviour of the species, local weather and local geography (Polacheck 1989). The appropriate design also depends on the amount of prior knowledge that exists about the distribution and abundance of the population. The ‘simple random survey’ design involves the random allocation of cruise tracks within each stratum so that each area has the same probability of being surveyed. The ‘variable coverage probability design’ is useful if conditions in strata are unpredictable, e.g.

[12] See Appendix VII cetacean distribution map

presence of a shifting ice-edge within a stratum. ‘Non-random survey designs’ involve the direct selection of areas that are known to have high cetacean densities. This can lead to positive bias in the results however and post-stratification is sometimes required (Hiby & Hammond 1989).

Counts of animals during surveys usually involve a minimum estimate resulting from a count of the maximum number of dorsal fins observed at any one time, unless the animals are individually identifiable to researchers (Hammond 1991). Cue-counting is an alternative to counting the actual number of animals sighted and involves estimating the number of animals present in an area by dividing the number of cues counted by the cue rate previously estimated. The most common cue used is the respiratory blow. By counting the number of blows observed on a survey and dividing this number by the blow rate for an individual, an approximation of the number of animals present can be achieved (Hiby & Hammond 1989).

All surveys techniques that rely on visual sightings are subject to bias from inter-observer variability, fatigue and variability in sighting conditions. While certain sighting conditions can be quantified, such as cloud cover and sea state, others are rarely estimated at all but may still affect the likelihood of sighting animals in the water. Hiby & Hammond (1989) suggest that other sighting conditions be quantified in further attempts to standardise the survey protocol and suggest, for example, the production of ‘sea colour charts’.

Inter-observer variation can be reduced by training, ensuring that observers have the opportunity to achieve a search image, that the survey protocol has been correctly interpreted, and the resulting data will therefore be viable (Øien 1996; Laake *et al.* 1998). Laake *et al.* (1998) found that inexperienced observers recorded only a quarter of the animals spotted by experienced observers during an aerial survey. Øien (1996) encountered problems with data viability in large-scale projects when local groups misinterpreted protocols.

Non-land-based surveys are conducted from small aircraft or from ships. The advantages of shipboard surveys are that they are generally unrestricted in their range whereas aircraft can be limited by fuel and airspace, boats can get close to cetacean pods (e.g. for photography) whereas aircraft cannot, and they are less likely to lose cetaceans, once sighted in ideal conditions (Hiby & Hammond 1989). The disadvantages are that cetaceans may actively avoid boats or may be attracted to them, due to engine noise or their physical presence, as was shown by Leutkebohle (1995) and Würsig *et al.* (1998). Also, estimating the distance of cetaceans from boats is difficult without sophisticated equipment (Hiby & Hammond 1989). The advantages of aircraft surveys are thus that cetaceans are unlikely to make efforts to avoid the approaching craft, that large areas can be covered in a single survey, and that photography can be used to determine pod size and even animal size, following calibration with a standard measurement. However, aerial surveys are restricted by cloud cover and sun glare in addition to wind speed, direction and strength and they require a Perspex nose bubble or 'belly-viewing port' in order that downward observations can be made (Hiby & Hammond 1989).

Studies have been conducted using aerial, land and ship-based surveys simultaneously to determine which method gave the best results (Kraus *et al* 1983; Laake *et al* 1998). Kraus *et al.* (1983) concluded that aerial surveys consistently sighted only 10 – 20% of the groups in the area as sighted from land. Laake *et al* (1998) noted that an air survey recorded only 35% of the groups present and commented that such surveys generally always underestimated cetacean abundance, due to perception and availability biases. Perception biases are caused by the visibility factors that limit observations over and above those that limit ship and land surveys. Availability biases are associated with the speed at which a plane travels in relation to the period of time animals remain submerged: animals may remain underwater as the plane passes over them. Forney *et al* (1991) concluded that aerial surveys were best reserved for relative abundance estimates. Although Würsig *et al.*'s (1998) work demonstrates that cetaceans react to boats, most authors agree that shipboard surveys are the superior method (Kraus *et al* 1983; Forney *et al* 1991).

The most comprehensive population survey carried out in European waters to date was the SCANS survey (Hammond *et al* 1995). Shipboard and aerial surveys were carried out along line transects to estimate numbers for all cetacean species in the North Sea, Danish waters, Swedish waters, the Celtic Sea and the English Channel. The estimate of abundance of harbour porpoises for the entire survey area was 352,523 (CV = 0.14). The SCANS estimates, together with more recent results, are summarised in Table 2.

2.5.2. North Sea, Baltic Sea and Swedish coastal waters

An annual by-catch of 4,449 porpoises was estimated to occur in the North Sea Danish turbot and cod fisheries alone (Vinther 1995). With a SCANS abundance estimate of 150,250 porpoises for the four strata included in this observer study, this by-catch estimate amounts to 3.1% of the population in the area (Hammond *et al*. 1995). Further, the annual removal rate in these Danish fisheries amounts to 1.7% of the entire North Sea harbour porpoise population as estimated in the SCANS survey (Vinther 1995).

In Swedish waters, Carlström & Berggren (1996) estimated a by-catch rate of 36 porpoises per 10,000 net km*hr which amounted to an annual removal rate of 2.9% of the harbour porpoise population in the Skagerrak Sea. These removal rates both exceed the 1% threshold advised by the IWC (1995).

The population of harbour porpoises in Danish waters was estimated at 67,490 by the SCANS project in 1994 (Hammond *et al.* 1995) and to exceed 100,000 by Teilmann & Lowry (1996). Nonetheless, the North Sea and Baltic Sea harbour porpoise populations are vulnerable to high by-catch rates. They are considered to be separate stocks (Teilmann & Lowry 1996) and both warrant protection under Appendix II of the Convention for the Conservation of Migratory Species of Wild Animals (Bonn) Agreement.

The harbour porpoise is reported to be in serious decline in the Baltic Sea, Danish waters (Berggren & Pettersson 1989; Lindstedt & Lindstedt 1989; Kock & Benke 1995, 1996) and in the northern North Sea (Evans *et al.* 1993), where by-catch rates are higher than in any other part of the North Sea (Vinther 1995). The spring migration of porpoises into the Baltic Sea, and that in the winter back to the North Sea, has now ceased (Anderson 1982; Kinze 1985; Clausen & Anderson 1988) and the species was not sighted in the southern North Sea during the SCANS surveys (Hammond *et al.* 1995).

The age and sex composition of porpoise by-catch in the North Sea, the Baltic Sea and Swedish waters varies. In the North Sea and off the West Coast of Scotland, of 31 by-caught porpoises examined by Northridge & Hammond (1999), 61.3% were female. A predominance of females was also shown in a study of porpoise by-catch in German waters (Siebert *et al.* 1993). However, other studies in Denmark and Scotland have recorded a higher proportion of males in by-catches (Clausen & Anderson 1988; Benke *et al.* 1991; Kinze, 1994; Pierce & Santos 2000). In Danish waters, a carcass salvage program and a reporting scheme involving one fishing vessel both suggested that more males were by-caught than females (Kinze 1994). Note however that the numbers involved in these studies tend to be small: the sex ratio observed by Northridge & Hammond (1999) did not differ significantly from 1:1.

Several authors comment on the predominance of calves and juveniles in the by-catch (Benke *et al.* 1991; Kinze 1994). Kock & Benke (1995, 1996) do not expressly mention the proportion of calves in the by-catch but do record an increase in by-catch rates between August and November inclusively, when neonates are most common in the population. These data may indicate differences in the distribution of porpoises when nursing very young animals.

All the studies cited above refer almost exclusively to the by-catch of harbour porpoises. Although there is some evidence that by-catch includes a disproportionately high number of males and juvenile porpoises, larger sample sizes are required before any definitive conclusion could be drawn.

2.5.3. Celtic Sea, Bay of Biscay and the Western Approaches

Reported cetacean by-catch rates are particularly high in the Celtic Sea, a very productive fishing ground supporting numerous fisheries and inhabited by numerous cetacean species. However, the high by-catch rate per unit of fishing effort is not simply due to the high number of cetaceans in the area. From results of a stranding reporting scheme started in 1994, Rogan & Berrow (1996) estimated that 5.5% of the harbour porpoise population in region (SCANS estimate) were lost annually to by-catch in the set gillnet fisheries. This was substantiated by an observer study, which indicated that 6.3% of the population were removed annually by incidental capture in the British and Irish hake gillnet fisheries alone (Tregenza *et al.* 1997a). The increase noted from one study to the other may represent inter-annual variation either in fishing effort or cetacean distribution, or may be an artefact of the different data collection protocols. Either way, both removal rates are very large, highlighting the fact that by-catch of harbour porpoises in the Celtic Sea is unsustainable. Of particular concern is the fact that these data refer only to by-catch in gillnet fisheries for certain species and, in a subsequent paper, Tregenza & Collet (1998) highlighted the need for further investigation into the impact of mid-water pelagic trawls and the mackerel, pilchard and anchovy fisheries. If Carlström & Berggren's (1996) model of porpoise population growth holds true for all stocks and the net growth rate is thus 4%, these removal rates are alarming and demonstrate the urgent need for by-catch reduction measures in this fishing ground.

There is also high mortality of common dolphins in mid-water trawl fisheries in the Celtic Sea. Goujon *et al.* (1994) estimated that 1.5% and 3% of common and striped dolphin populations respectively were taken annually in the French tuna driftnet fishery which concludes its fishing season in the southern Celtic Sea (Collet *et al.* 1992). Tregenza & Collet (1998) report on an observer study of mid-water trawl fisheries in the Celtic Sea and, although the study generated insufficient data to estimate annual by-catch, they expressed the belief that the mortality rate in trawls was high. These fisheries may have caused the deaths of many of the 112 common dolphins stranded on the SW coast of England in 1992 (Kuiken *et al.* 1994).

Data on the age and sex composition of cetacean by-catches in the Celtic Sea and the Bay of Biscay are scarce. In the mass stranding of common dolphins on the Brittany coast in 1997, 60% of the diagnosed by-catch mortalities were of males (Tregenza & Collet 1998). In the French tuna driftnet fishery, Goujon *et al.* (1993) found that half the by-catch of common and striped dolphins comprised adult females and nearly half comprised calves and neonates. In subsequent studies of the same fishery, the dolphin by-catch comprised mostly young animals (Goujon *et al.* 1994). On occasions when no calves were caught, however, the by-catch comprised mainly males (Goujon *et al.* 1993). In a study of mid-water pelagic trawls, Morizur *et al.* (1999) found that the catch of common and white-sided dolphins consisted entirely of adults.While it is possible that this may be related to the behaviour of the animals at the time of fishing (Berrow pers.comm), it may also be concluded from these disparate data that dolphins segregate by age and sex and which group is by-caught thus depends on the timing and location of the fishery.

2.5.4. Mediterranean

Very few data exist on the impact of by-catch on cetacean populations in the Mediterranean. This is not only because there are few by-catch estimates but also because population sizes are mostly unknown. Silvani *et al.* 1999 estimated the common dolphin population size to be 14,736 ± 5,894. Estimates for striped dolphins vary from 17,728 ± 5,850 to 100,000 (Notarbartolo di Sciara 1994; Cognetti 1995; Silvani *et al.* 1999). These estimates refer to relatively small areas, in which the only data are from the 27 Spanish driftnet vessels (Jefferson & Curry 1994; Silvani *et al.* 1999). These boats alone were estimated to remove 1% of the common and striped dolphin populations annually prior to cessation of their activity.

Silvani *et al.* (1999) provide the only analysis of the composition of the cetacean by-catches in the Mediterranean, again based on the study of the Spanish driftnet fishery. They found that 11% of by-caught common dolphins were adults, with males and females equally represented, while the rest were calves or juveniles. Similarly, only 19% of by-caught striped dolphins were adults, although 76% of the catch consisted of males and the only sexually mature animals were all male.

2.5.5. Gulf of Maine and Bay of Fundy

Amendments to the MMPA in 1994 required population assessments be carried out annually on all strategic marine mammal stocks (Caswell *et al.* 1996). Consequently, available by-catch estimates for harbour porpoises in the Gulf of Maine (Bravington & Bisack 1996; Bisack 1997) the Bay of Fundy (Trippel *et al.* 1996) can be extrapolated to estimates the impact on the population.

In recognition that data leading to abundance estimates are usually fragmentary and/or taken from different sources, Caswell *et al.* (1996, 1998) applied a Monte Carlo Uncertainty Analysis Model to the data available for the region. They concluded that, despite uncertainties in the population estimates, it was likely that the anthropogenic removal rate exceeded the net growth rate of the porpoise populations and there was thus cause for concern. Despite the uncertainties, the various abundance estimates for porpoises in the Gulf of Maine and Bay of Fundy are rather similar. Impact estimates range from 2.5% to 6.1% of the population being by-caught annually (Waring *et al.* 1990; Bravington & Bisack 1996; Caswell *et al.* 1996; Trippel *et al.* 1996; Bisack 1997). Using the mean annual by-catch estimate and the most recent population estimate available (Bisack 1997), it was calculated that, on average, 3.3% of the porpoise population in the Gulf of Maine and the Bay of Fundy is lost to by-catch each year.

The impact of by-catch on harbour porpoises has generally declined over the last few years. Using a population estimate of 47,200 porpoises, Caswell *et al.* (1996) show that the removal rates for 1991 to 1994 were 6.1%, 4.2%, 2.5% and 2.9% respectively. Assuming the same population size, and Bisack's (1997) by-catch estimate for 1995, we calculate that the by-catch rate in 1995 remained at 2.9%. Despite the downward trend, these figures still exceed the removal threshold of 1% of the population and justify the reduction measures that are being implemented by the NMFS under guidelines provided by the MMPA.

Few published data are available about the composition of by-catches in the NW Atlantic. Many authors mention that by-caught harbour porpoises and common dolphins are usually yearlings or are at least prepubescent (Smith *et al.* 1983; Waring *et al.* 1990; Cox *et al.* 1998). Waring *et al.* (1990) mentions that 72% of pilot whales were caught by the DWFs. Fontaine *et al.* (1994) noted that, in the Gulf of St. Lawrence, 58% of the sexually mature harbour porpoise females caught were lactating and concluded that the actual death toll due to by-catch should be increased by a standardised correction factor to account for the loss of calves orphaned by these females. On examination of stranded harbour porpoises Cox *et al.* (1998) concluded that all by-caught animals were immature and considered this to be evidence for age-segregated distribution, with younger animals staying closer to shore than adults.

2.5.6. Conclusion

In conclusion, for areas where there are data on by-catch rates, there is usually at least one paper that expresses the rate of removal as a percentage of the overall population size. This can then be compared with the percentage by which the population grows each year. The removal rates range from 1.5% of the common dolphin population in the French tuna driftnet fishery (Goujon *et al.* 1993) to 6.3% of the harbour porpoise population in the hake gillnet fishery in the Celtic Sea (Tregenza *et al.* 1997a). All the estimates quoted exceed the 1% threshold recommended by the IWC (IWC, 1995) and most of these data represent the take from only one fishery in areas where many other fisheries are active. The data are thus alarming and there can be no dispute that such unsustainable by-catch requires urgent action.

The wide variation observed in the catch composition between studies may suggest that by-catch is indiscriminate and that no one sector of the population is more vulnerable than the other. However, if this were the case, one would perhaps expect to find a more even distribution of each demographic category within each fishery and within each fishing season but this does not seem to be the case. In fact, most studies indicate that a particular demographic sector, either young animals, females, males or adults is caught at any one time (Benke *et al.* 1991; Collet *et al.* 1992; Goujon *et al.* 1993; Siebert *et al.* 1993; Kinze 1994; Lowry & Teilmann 1994; Iwasaki & Kasuya 1996; Tregenza & Collet, 1998; Morizur *et al.* 1999; Silvani *et al.* 1999). Thus it is likely that by-catch is essentially indiscriminate. Variation in the catch composition reflects the sector of the cetacean population predominant in the area at the time of the fishing (Waring *et al.* 1990; Jefferson & Curry 1994; Cox *et al* 1998; Karchmarski, 2000) rather than the selectivity of the nets. These data are consistent with segregation of herds according to age and sex as noted in bottlenose dolphins and harbour porpoises (Waring *et al.* 1990; Cox *et al.* 1998).

Nearly all porpoises caught in herring weirs are yearlings and Read & Gaskin (1990) suggested that this probably occurs because the younger animals often stay alone in the Gulf of Maine to overwinter. Cockcroft (1994b) carried out detailed analyses of the events surrounding the by-catch of cetaceans in shark nets off Natal in South Africa. In collecting environmental, physiographical and biological data from each by-catch event, he aimed to find any common factors. The only environmental data that correlated significantly with by-catch were the mean water temperature and the direction of the prevailing current. Analyses of the biological data matrices revealed that the by-catch of bottlenose dolphins consisted mainly of sucklings and lactating females, most of the indo-pacific humpbacked dolphins entangled were large males, and that more female and juvenile common dolphins were caught than males (Cockcroft 1994b). He concluded that, apart from changes in the current affecting the actual by-catch rates, the composition of the catch depended only on which sector of each population used the area most. If the by-catch consisted of mainly females and young calves, it was probable that the area was used by these animals as a nursery ground, likewise, the absence of females in an area may indicate that it is used by feeding bachelor groups (Cockcroft 1994b).

Iwasaki & Kasuya (1996) considered female Pacific white-sided and northern right whale dolphins (*Lissodelphis borealis*) to be more vulnerable than males while De Haan *et al.* (1997) argue that by-catch may occur more in the breeding season because animals are more easily distracted. It may also be the case that younger animals are more vulnerable to entanglement in nets, either through lack of caution, a lack of skill (Smith *et al.* 1983; Fertl & Leatherwood 1997), exuberance, curiosity and distraction (Kastelein *et al.* 1995b) or boldness in the presence of adults (Kastelein *et al.* 1995a).

The consequence of certain population components being more vulnerable than others is a very important consideration. Disproportionately high numbers of calves and reproductively active females in the by-catch are of particular concern. Death of calves may also result when lactating females are caught - such that Fontaine *et al.* (1994) added 13% to their estimated death toll to account for death of calves when their mothers were caught. The reduction in population growth rate due to loss of reproductively active females will exacerbate the direct damage caused by by-catch. Demographic changes resulting from declining population size are already apparent in the NW Atlantic and in Danish waters, where female harbour porpoises reach sexual maturity at a significantly younger age than was the case prior to 1973 and calves are generally larger (Clausen & Anderson 1988; Read & Gaskin 1990; Read, 1994).

As a general conclusion, the published data on cetacean by-catches indicate the need to introduce by-catch reduction measures in European waters, perhaps following the US model (driven by the provisions of the MMPA).

The importance of being able to predict the impact of by-catch on cetacean populations is clear when attempting to determine the need, or the urgency of need, for action in given areas. It must be admitted that, in many areas, the data needed to make predictions are lacking. Funding bodies need to be convinced that their money is going to be well-spent, and this may be difficult where good data on by-catch rates or population sizes are lacking.

However, lack of data should not be allowed to justify procrastination with respect to plans for by-catch reduction. The studies conducted so far have shown that, in some areas, as much as 6.3% of the harbour porpoise population is being removed annually in by-catch in a single fishery. Such levels of mortality are clearly unacceptable, even without taking into consideration the (less easily quantified) negative impacts of other anthropogenic impacts such as pollution, increasing boat traffic and habitat destruction. The need for action is particularly pressing where analysis draws attention to a disproportionate death rate in calves and females, which may indicate fishing in nursery areas or during breeding seasons.

Understanding the impact of by-catch is imperative both to identify where problems exist in the first place, and to monitor the long-term success of by-catch reduction measures in achieving sustainable removal rates. However, the necessary data should be collected in tandem with efforts to reduce by-catch and the existence of such data should not be considered as a necessary pre-requisite to implementing by-catch reduction schemes.